ISWS/CIR-138/79

Circular 138

STATE OF ILLINOIS

ILLINOIS INSTITUTE OF NATURAL RESOURCES

Effect of Agriculture on Cedar Lake Water Quality

by DONALD P. ROSEBOOM, RALPH L. EVANS, and THOMAS E. HILL

ILLINOIS STATE WATER SURVEY

URBANA

1979

CONTENTS

Printed by authority of the State of Illinois
17711 (12-79-1000)

Effect of Agriculture on Cedar Lake Water Quality

Donald P. Roseboom, Ralph L Evans, and Thomas E. Hill

ABSTRACT

This study was made to delineate the type, distribution, and magnitude of orchard-related pesticides in the aquatic environment of Cedar Lake which includes the older impoundments of Little Cedar Lake and the Alto Pass Reservoir. Cedar Lake is a source of water supply for Carbondale, Illinois. The study included analyses for mineral and pesticide content of the soils, the water and bottom muds of the three lakes, and the water and sediments of the four major tributaries. In addition, the general limnology and biological characteristics of Cedar Lake were defined, sedimentation surveys of the two older lakes were performed, and relevant data regarding pesticide content in fish were evaluated.

The results show that application of pesticides in well-sodded orchards does not pose a serious threat to the water quality of Cedar Lake, but a threat does exist when such orchard lands are converted to row crops, subdivisions, or other uses that destroy the sod cover that minimizes soil erosion. Pesticide residues that make their way by soil movement to the aquatic environment do accumulate in the bottom muds of the lakes, and may become solubilized and thus transferred to the lake waters.

The waters of Cedar Lake stratify in summer resulting in depletion of dissolved oxygen in the lower 15 feet of water. Lake management to minimize such oxygen depletion is needed to control taste and odors and the introduction of undesirable metals in the water supply. The sedimentation surveys showed that soil loss on the watershed is excessive and that soil erosion prevention procedures are needed.

The desirable game fish in the lakes contain mercury concentrations in excess of FDA tolerance limits despite the fact that mercury in excess of background levels was not found in the soils, suspended sediment, nor lake bottom sediments in the watershed. Research as to possible effects of inundated vegetation on the transfer of mercury from water to fish is suggested. DDT and dieldrin in fish are not a cause for concern in the Cedar Lake watershed.

INTRODUCTION

Cedar Lake, a man-made impoundment in Jackson County, Illinois, is located approximately 6 miles south of the city of Carbondale west of Illinois Route 51. The lake is a joint undertaking between the city of Carbondale and the U.S. Forest Service. Its 30-mile long shoreline is in public ownership with the city owning and managing the northern half and Shawnee National Forest (U.S. Forest Service) owning and managing the southern half of the lake.

At normal pool elevation (431.0 feet mean sea level), the surface area of Cedar Lake is about 1800 acres and its volume is about 41,500 acre-feet. Dam closure was completed in December 1973 and it reached normal pool during the winter of 1974. The dams consist of a main dam (figure 1) and a saddle dam (figure 2). The lake's primary purpose is to provide a long range water supply for the city of Carbondale. In addition, its waters are readily available for recreational opportunities.

Watershed

The 30.2-square-mile watershed of Cedar Lake is located entirely within the western part of the Shawnee Hills section of the Interior Low Plateau province. As shown in figure 3, the watershed extends into southern Jackson and northern Union Counties. The village of Alto Pass is near the southwest border. Massive cliff-forming sandstone formations outcrop in the steep valley walls. Figures 4 and 5 depict typical outcroppings along the shoreline of Cedar Lake.

Approximately 50 percent of the watershed area is in woodland, 13 percent is in apple and peach orchards, and 12 percent is in row crops. The remaining land is in permanent pasture, meadow, and small grain fields. Soils in the watershed are silt loams consisting principally of Alford, Hosmer, Neotoma, and Wellston types. Orchard soils range in organic content from 2.6 to 4.7 percent based upon weight loss on ignition at 800° C.

In Illinois there has been an annual reduction in land acreage supporting orchards of about 2.9 percent during the 7-year period 1968-1975. During this time, the reported losses in apple and peach trees were 10 and 23 percent, respectively. Orchard land has been converted to row crops, pasture, subdivisions, and other seemingly more profitable

Figure 1. Main dam on Cedar Lake

Figure 2. Saddle dam on Cedar Lake

uses. Such conversions other than for pasture will destroy the continuous sod cover common in orchard lands of the Cedar Lake watershed that minimizes soil erosion.

The major tributaries to Cedar Lake are Poplar Camp, Clay Lick, and Mill Creeks, and the East and West Branches of Cedar Creek (see figure 3). From the valleys of the creeks, three prominent legs are formed on the lake. The water intake structure for the city of Carbondale is located on the Poplar Camp leg and is shown in figure 6. The average streamflow to the lake system is about 14 cubic feet per second (cfs).

Other lakes on the watershed include Little Cedar Lake and the Alto Pass Reservoir. Little Cedar Lake, formed in 1969, is located south of Cedar Lake on the Cedar Creek leg (see figure 3). It is entirely owned and managed by the U.S. Forest Service. No development is planned for the lake since it is to be maintained as a 'quiet natural area.' Its unique spillway, consisting of an outcrop formation, is shown in figure 7.

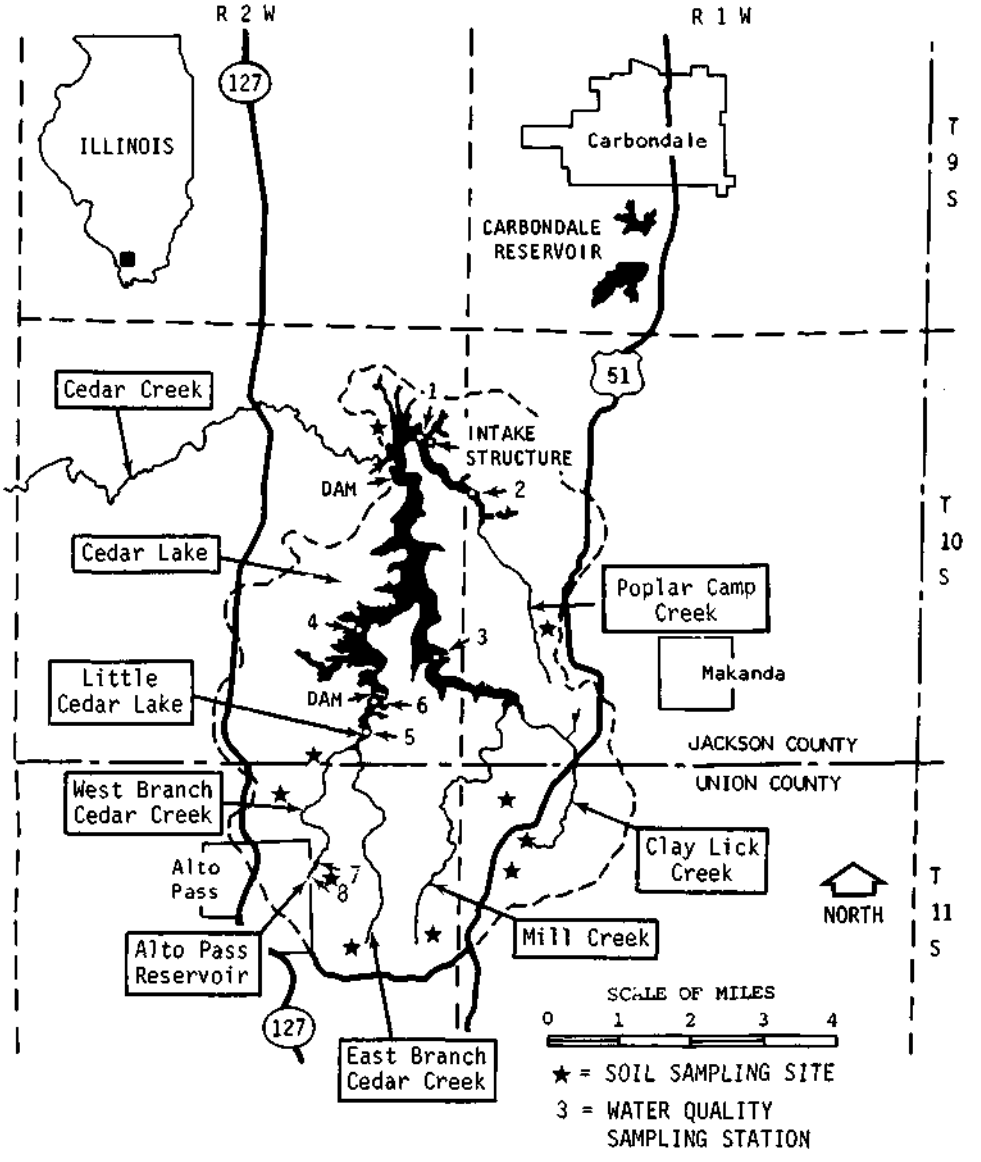

Figure 3. Cedar Lake watershed

The Alto Pass Reservoir, formed in 1967, serves as a source of public water supply for the village of Alto Pass. It is located on the upper end of the West Branch of Cedar Creek as shown in figure 3.

Objective and Scope

The Cedar Lake study was performed for the Greater Egypt Regional Planning and Development Commission in conjunction with the Commission's Areawide Waste Treatment Management Plan, and was financed under Public Law 92-500, Section 208 of the 1972 Water Pollution Control Act Amendments. The study was carried out by the Water Quality Section of the Illinois State Water Survey. The field work was done in 1976 and the analyses were completed in 1977. The results were originally published in *Water Quality in Cedar Lake and the Relevance of Its Watershed* (Roseboom et al, 1978).

The principal objective of the study was to delineate the type, distribution, and magnitude of pesticides in the aquatic environment of Cedar Lake. The work was prompted by concern for the entrance into the aquatic environment of substances being employed in apple and peach orchards for pest control. It is anticipated that the findings will provide

Figure 4. Typical sandstone outcroppings along Cedar Lake shoreline

Figure 5. Typical sandstone outcroppings along Cedar Lake shoreline

the basis for decisions toward maintaining a water quality in the lake system satisfactory for its intended uses.

As part of the study, analyses for mineral and pesticide content were performed on: 1) the soils of orchards, row crops, pastures, and woodlands; 2) the water and bottom muds of Cedar Lake, Little Cedar Lake, and Alto Pass Reservoir; and 3) the waters, alluvial deposits, and suspended sediments of the four major tributary streams. Surveys to determine the extent of sediment accumulations were performed on the two older lake systems, and relevant fish data regarding pesticide content were evaluated. In addition, the algae and benthic organisms of Cedar Lake were examined and the general limnology of the lake was defined. Arsenic determinations were limited to soil samples.

Complete tabulations of the various observations and counts are available at the Water Quality Section (Box 717, Peoria, Illinois 61601) in a *Cedar Lake Open File Data Supplement.*

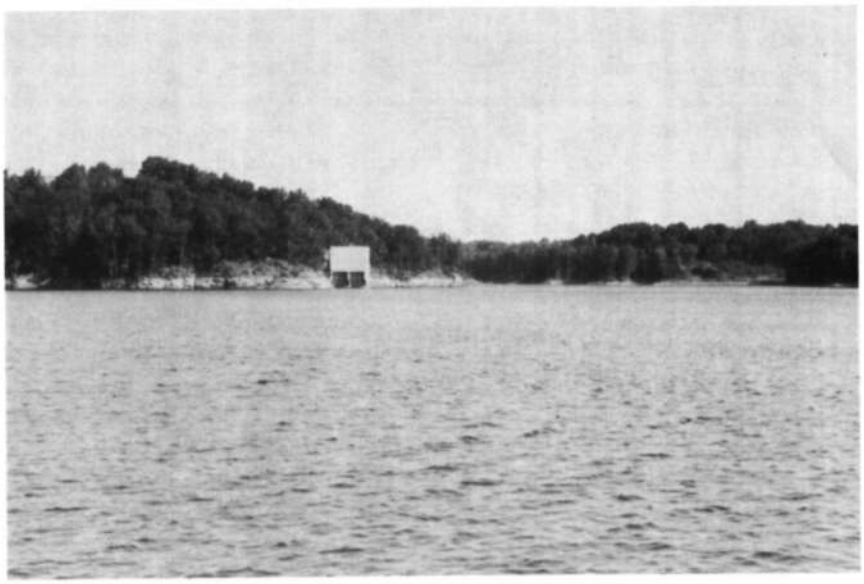

Figure 6. Water intake structure on Poplar Camp leg of Cedar Lake

Figure 7. Spillway for Little Cedar Lake

Acknowledgments

This investigation, sponsored and financially supported by the Greater Egypt Regional Planning and Development Commission, was conducted under the general supervision and guidance of Dr. William C. Ackermann, Chief of the Illinois State Water Survey. Illustrations were prepared by William Motherway, Jr., under the supervision of John W. Brother, Jr.; Mrs. Marilyn J. Innes prepared the camera copy; Mrs. J. Loreena Ivens and Mrs. Patricia A. Motherway edited the final report.

The authors are indebted to many persons for technical assistance, guidance, and advice during the course of this undertaking. Mr. Ike Kirkikis of the Greater Egypt Regional Planning and Development Commission was especially helpful in arranging for contacts at the local level and keeping us informed of other projects in the area. Charles Hendricks, Mikeal Jones, and George Lyons of the U.S. Forest Service provided advice on lake access and offered equipment. From the city of Carbondale Bill Boyd, Director of Public Works, Tom Harris, Manager of Lake Facilities, and Jim

Mayhugh, Superintendent of Water and Sewage provided maps and equipment that facilitated field sampling. James Sweitzer, President of the village of Alto Pass, arranged for information on the Alto Pass Reservoir. Kermit Lawson, Joe Barkley, Paul Ice, Clifford Miles, Bruce Currie, and Lewis Duncan, all of the U.S. Soil Conservation Service, provided soil maps, offered advice on soil sampling, and loaned equipment for that purpose. Harold Hunzicker (Union County) and Robert Frank (Jackson County) of the University of Illinois Cooperative Extension Service arranged for meetings with orchard owners.

Among the orchard owners who gave freely their time and advice, we are particularly indebted to Dan McQuire, Bill Flamm, C. Braden, Jim Eckert, Ray Grammer, James Hartline, James Rendleman, and Glen Draper. James Allen and Donald Garver of the Illinois Department of Conservation collected the fish samples and Robert Hite of the Illinois Environmental Protection Agency compiled fish-pesticide data and made it available. Dennis Stover, also of that agency, provided advice on water treatment practices.

Ronald Meyer, of the Illinois Natural History Survey, made available his copies of transactions of the Illinois State Horticulture Society. James Hundley of the Illinois Department of Public Health made arrangements for the organic pesticide analyses of sediments and soils and Frank Amore, former Water Survey Chemist, performed the analyses. Philip Hopke of the Environmental Research Analytical Laboratory at the University of Illinois developed the data on arsenic content of soils and Dave Hullinger of the Water Survey (Water Quality Section) performed analyses for all other heavy metals. David L. Gross and Jerry A. Lineback of the Illinois Geological Survey performed a reconnaissance of major outcropping formations of the watershed and offered advice on mercury content and sources.

SAMPLE COLLECTION AND LABORATORY PROCEDURES

The samples collected and the analyses performed varied considerably during the course of the study. *In situ* measurements were made on Cedar Lake and Little Cedar Lake. Water samples were gathered from both lakes and examined for chemical constituents, phytoplankton, and benthic organisms. Mud-water interface samples were examined for heavy metals as well as chemical constituents. On one date interstitial waters were collected for heavy metals determinations. On one date, also, the major chemical constituents of water obtained from the major tributary streams were ascertained as well as the heavy metals content of the suspended solids in the stream waters.

Samples of soil (obtained from land supporting orchards, row crops, pastures, and woodlands), lake muds, and alluvial deposits were examined for inorganic and organic pesticide residues. The following outlines the extent of sampling and the procedures used for collecting samples and performing analyses.

Water Transparency, Dissolved Oxygen, and Temperature

Field trips were made to perform *in situ* observations for water transparency, dissolved oxygen, and temperature on Cedar Lake and Little Cedar Lake. Observations were made on Cedar Lake at four sampling stations on nine occasions and on Little Cedar Lake at two sampling stations on seven occasions. The Cedar Lake stations were established near the water intake and on the three legs of the lake, i.e., Poplar Camp, Clay Lick, and Cedar Creeks. The two stations on Little Cedar Lake were located near the dam and upstream. The locations of the sampling stations are shown in figure 3.

Transparency. An 8-inch diameter Secchi disc with black and white quadrant markings attached to a calibrated line was used to measure water transparencies. The disc was lowered until it disappeared from view, lowered farther, and then raised slowly until it reappeared. The depth of each immersion was noted and the average of the two observations was recorded as the Secchi disc reading.

Dissolved Oxygen and Temperature. In situ dissolved oxygen and temperature measurements were made by use of a galvanic cell oxygen analyzer equipped with a thermister. An oxygen meter, Yellow Spring Instrument Company model 54 with a 50-foot cable, was used for this purpose. At the beginning of each day of measurement, the probe was standardized in lake water in which the dissolved oxygen content was determined by a modified Winkler Method as outlined by the American Public Health Association (1971). Temperature and dissolved oxygen measurements were obtained at 2-foot intervals starting from the water surface of the lake.

Water Chemistry

Water samples were collected from the four stations on Cedar Lake and the two stations on Little Cedar Lake. Samples were collected at the surface, about 3 feet from the lake bottom, and at the mud-water interface on five occasions. The time of collection represented early spring, early summer, late summer, early autumn, and late autumn. On

Table 1. Analytical Procedures

pH	Glass electrode method using Leeds and Northrup 7401 and Beckman 4500
Alkalinity	Potentiometric method
Hardness	EDTA titrimetric method
Nitrate–N	Chromotrophic method (West and Ramachandran, 1966)
Ammonia–N	Modified phenate method
Total silica	Heterophy blue method
Total phosphorus	Sample was digested with sulfuric-nitric acids mixture and determined by ascorbic acid method
Dissolved orthophosphorus	Ascorbic acid method after filtration through 0.45 μm filter paper
Algal growth potential	Method outlined by Wang et al. (1973)

Note: Unless otherwise stated all methods used were performed according to Standard Methods (American Public Health Association, 1971)

one occasion, water samples were collected from the tributary streams West Cedar, East Cedar, Clay Lick, Mill, and Poplar Camp Creeks. Also, on one occasion the interstitial waters from the lake bottom muds at the stations on Cedar Lake and Little Cedar Lake were obtained for analyses.

Collection. Lake water samples from the water surface and 3 feet from the bottom were obtained with a Kemmerer sampler. They were placed in plastic bottles, stored on ice until transported to the laboratory, and refrigerated until chemical analyses were performed. Water samples from tributary streams were collected in a plastic bucket, placed in plastic bottles, and handled thereafter the same as the lake water samples. Mud-water interface samples were collected about 3 centimeters (cm) above the lake bottom with a device developed by Sullivan (1967). The samples to be examined for heavy metals were acidified with nitric acid. The samples upon which chemical analyses were to be performed were handled in the manner described for lake water samples. Interstitial water samples, those samples derived solely from lake muds and sometimes called 'pore water,' were obtained from the supernatant of lake mud samples after about 48 hours of storage in plastic bottles.

Samples to be examined for ammonia determinations received special handling in the field. A 50-milliliter (ml) portion of the water samples was filtered through type HA, 0.45 μ millipore filters 37 mm in diameter. The filters were placed on filter pads which were held between two-piece circular holders. Positive pressure for filtering the samples was provided by a syringe to force the sample through the filters. The filtrates were collected in small plastic bottles. Micropore filtration eliminates any bacterial activity which could alter the ammonia concentration in the collected samples. This method of sample preservation is considered superior to acidification or other chemical additives.

Analyses. Chemical analyses on lake water and mud-water interface samples were made for pH, alkalinity, hardness, nitrate-N, ammonia-N, total silica, total phosphorus, dissolved orthophosphorus, and algal growth potential. The analyses were performed in accordance with the procedures outlined in table 1. The same determinations were performed on stream water samples with the exception of the ammonia-N analysis. Results are expressed in milligrams per liter (mg/1) except in the case of temperature and pH. Temperature is expressed in Celsius units and pH is dimensionless.

Interstitial water samples from six stations were examined on one occasion for the heavy metals lead, copper, zinc, iron, and mercury. Interface samples were collected and analyzed for the same heavy metals on four dates representing spring turnover, two summer stratification periods, and autumn turnover conditions. The procedures used for heavy metals determinations are outlined in the section on Pesticides.

Algal growth potential (AGP) was determined for the three strata of lake water. Algal growth potential is a laboratory procedure to evaluate the ability of filtered water samples to sustain algal growth, under standardized laboratory conditions, following inoculation with mixed algal culture (Wang et al., 1973). It is defined as the algal or organic mass resulting from a 7-day incubation of an algal culture grown on a natural substrate under standardized light and mixing conditions and expressed as milligrams of dry organic mass per liter of sample.

Algae

Algal collections were made at the water surface of Cedar and Little Cedar Lakes at the stations depicted in figure 3. Collections were made during the months of April,

June, July, August, September, and October. Altogether 34 samples were collected and examined.

Collections. Surface water samples were collected by immersing a small-mouth glass bottle, containing formalin as a preservative, until a sample volume of 380 ml was attained. The samples were capped and stored at room temperature until examined.

Examinations. Examinations were generally performed within 2 weeks, at which time the sample was thoroughly mixed and a 1-ml aliquot pipetted into a Sedgwick-Rafter cell. An inverted phase contrast microscope equipped with 10X eyepieces, 20X objective, and a Whipple disc was used for identification and counting purposes. Five short strips (about 280 fields) were counted. The algae were identified to species using several keys (Palmer, 1959; Patrick and Reimer, 1966; Prescott, 1962, 1970; Smith, 1950; Tiffany and Britton, 1951). They were classified in four main groups: blue-greens, greens, diatoms, and flagellates.

Benthic Organisms

The four stations on Cedar Lake and the two stations on Little Cedar Lake were sampled for benthic organisms also. In addition, two stations on the Alto Pass Reservoir were sampled for benthos. Sixteen samples were obtained from Cedar Lake, six from Little Cedar Lake, and two from the Alto Pass Reservoir.

Collections. Benthic organisms were obtained from Cedar Lake on April 14, May 12, July 29, and October 28. On May 12 each sample consisted of three ponar dredge grabs. The other three samples consisted of three 6x6-inch Ekman dredge grabs each. The Little Cedar Lake stations were sampled on April 14, July 29, and October 28. All samples consisted of three Ekman dredge grabs. The Alto Pass Reservoir stations were sampled August 11 and consisted of three Ekman dredge grabs. All samples were washed in the field through a U.S. Standard No. 30 mesh seive bucket, placed in quart jars, and preserved with 95 percent ethanol.

Examinations. In the laboratory the organisms picked from the bottom detritus were counted, identified, and preserved in 70 percent ethanol. Sample dry weights were obtained by drying each sample for 24 hours at 60 C.

Soils, Lake Muds, Alluvial Deposits, and Suspended Sediments

Soil samples were collected from 10 orchards, 6 row crop fields, 10 pastures, and 8 woodland parcels. Altogether, 760 soil samples were gathered. However, only composited samples were anlayzed. Lake muds were collected from Cedar Lake, Little Cedar Lake, and Alto Pass Reservoir. Alluvial deposits were collected at the mouth of three major tributary streams in lieu of a suspended solids sampling program. Streamflow was high enough on one occation to justify sampling for suspended sediment content in five tributary streams. Determinations for heavy metals content were performed on the samples; organic pesticide analyses were similarly performed except on the suspended sediment obtained from stream water.

Collections. Samples of *soil* were obtained with a soil sampler manufactured by the Oakfield Apparatus Company. The sampler consists mainly of a stainless steel split sleeve coring body 3/4 inch in diameter and 12 inches long.

On each parcel of land selected for soil sampling, two diagonals were marked. The lengths of the diagonals for orchard and row crop land averaged about 800 linear feet; for pasture and woodland parcels the diagonals were more uniform and averaged about 1000 linear feet. On land supporting orchards and row crops, 16 soil sampling sites were established at equal intervals on each diagonal. Soil samples were obtained by pressing the soil sampler into the sampling site to a depth of 12 inches. Each 12-inch core of soil was divided into two 6-inch segments representing *top* and *bottom* segments of the core.

Along one diagonal, the 16 top 6-inch soil segments were placed together, as were the 16 bottom soil segments, to provide two composited samples along the diagonal. Along the other diagonal, each of the 6-inch segments was stored separately providing 32 discrete soil samples. The same procedure was used on pasture and woodland parcels except the discrete samples were limited to 5 sampling sites.

Those samples collected for compositing were first 'grated' through 1/4-inch diameter openings that had been drilled through the bottom of a stainless steel pan. The 'grated' particles collected in another stainless steel pan were then sifted through a stainless steel riffle compositor four times. Next, the soil samples were placed in polyethylene bags for heavy metals analyses, and in 6-ounce glass jars with aluminum foil covers capped by molded screw covers for organic pesticide analyses. These preparations were accomplished in the field. The samples to be examined for organic pesticide content were kept cool in an ice chest and upon delivery to the laboratory were frozen until analyses began.

Samples of *lake muds* were obtained by two methods. In one case, three grab samples were obtained with the Ekman dredge from four stations on Cedar Lake, two stations in Little Cedar Lake, and two stations in the Alto Pass Reservoir. These mud samples were also the source of interstitial water samples. In the other case, limited sampling was accomplished by a sediment sampler developed by the Water Survey, and included cores of lake mud from one site in Little Cedar Lake and two sites in the Alto Pass Reservoir.

Lake mud samples were placed in polypropylene containers and in quart glass jars with aluminum foil lined caps, respectively, for heavy metals and organic pesticide analyses. They were preserved and stored in the same manner as the soil samples.

The exposure of *alluvial deposits* at the mouth of three streams (Poplar Camp, Clay Lick, and Mill Creeks) permitted sampling by hand. Samples were stored and preserved as described for lake muds.

On one occasion, the water flow of five streams was relatively high and samples were taken principally for *suspended sediment* content. The samples were cooled immediately upon collection and maintained so until examined for heavy metals concentration on the suspended particles.

Analyses. Determinations for lead, copper, zinc, iron, and mercury were performed on samples of soil, lake muds, alluvial deposits, and suspended solids. Arsenic determinations were performed only on soil samples. A Perkin-Elmer 305 Atomic Absorption Spectrophotometer was used for lead, copper, zinc, iron, and mercury determinations. Arsenic levels in soils were ascertained by neutron activation techniques. The procedures for analyses by atomic absorption are described below.

Soil samples were air dried for at least 7 days. Samples of lake mud and alluvial deposits were dried in an oven at 60° C for 2 days. After being dried the samples were pulverized in an electric blender and passed through a 250 mesh per inch (63 μ m opening) sieve (U.S. Standard Testing Sieve No. 230) to remove all sand and larger particles. Portions of the silt-clay mixture (0.01 to 1.0 gram) retained on the sieve were weighed and analyzed. Determinations for lead, copper, zinc, and iron were made by digesting the sample with nitric acid and perchloric acid at temperatures below boiling (Peden, 1976), diluting to a constant volume, centrifuging, and aspirating the supernatant into the spectrophotometer. Duplicate samples, with known concentrations of the appropriate metal, were subjected to the same procedure for control purposes. Samples of mud-water interface and interstitial waters were examined similarly. Generally, a volume of 100 ml was used for the liquid samples that had been acidified in the field with nitric acid.

Determinations for mercury were made by digestion with nitric acid, 4 percent potassium dichromate, and 5 percent potassium disulfate at temperatures well below boiling. Samples were then reduced with a 10 percent mixture of $NH_2OH \cdot HCl$ and 10 percent tannous chloride. The cold vapor technique (Hatch and Ott, 1968) was then employed with the atomic absorption spectrophotometer. About 1 to 2 grams (g) of soil or sediment samples and 50 ml of liquid samples were analyzed in duplicate, often with the duplicate containing a low quantity of mercury.

The procedures used to determine the arsenic content in soil samples included neutron activation techniques. The following steps were used.

Approximately 0.500 g of soil was accurately weighed into a precleaned polyethylene vial. To prevent contamination, the vial was sealed in a larger vial. Two standards were prepared by pipetting 0.02 or 0.05 ml of a 996 μ g/ml standard solution of As (Spex Industries) onto a 5 cm disk of Whatman 42 filter paper in another polyethylene vial. A reference sample of National Bureau of Standards SRM 1633, trace elements in coal fly ash, was prepared in the same manner as the soil samples.

A set of approximately 20 samples, the fly ash sample, and the 20 and 50 μ l standards were simultaneously irradiated for 3 hours in the rotating sample holder of the Illinois Advanced TRIGA Reactor Facility operating at 700 kilowatts. The neutron flux is approximately 2×10^{12} n/cm^2-sec. The samples were permitted to decay for 2 days before beginning the counts. Each sample, the fly ash, and the standards were each counted for 4000 to 6000 seconds with a 10 percent efficiency Ge(Li) detector system having a resolution of 2.1 kiloelectron volt (keV) at 1333 keV. The data were accumulated in a 4096 channel pulse height analyzer operating in a real time mode. A precision pulse generator was used to provide live time correction. The data were recorded on computer-compatible magnetic tape and the spectral information was reduced automatically to mass concentrations by the computer code (Maney et al., 1976). Since duplicate standards were used and two relatively intense gamma-ray lines (559 and 657 keV) were observed in the decay of ^{76}As, four values were obtained for each sample. The mean value was obtained by a weighted averaging procedure using the statistical uncertainty of the counting data to weight the data points. The values reported are the averages of the mean. There do not appear to be any systematic errors since good agreement was achieved for the arsenic concentration of the NBS standard compared with the certified value of 61 ± 6 and the INAA round robin value (Ondov et al., 1975) of 58 ± 4. The detection limits for heavy metals are given in table 2.

Table 2. Detection Limits for Heavy Metals

	Percent recovery	*Detection limits (ppm)*
Lead, Pb	94.4	0.0003
Copper, Cu	96.3	0.0003
Zinc, Zn	100.3	0.0002
Iron, Fe	103.1	0.0002
Mercury, Hg	99.6	0.0085*
Arsenic, As		0.5000

*ppb

Analyses for organic pesticides were performed on samples of soil, lake muds, and alluvial sediments by gas chromatography. Detection limits are shown in table 3. The pesticides were extracted from the samples with alcohol-acetone, flooded out with water, and partitioned into petroleum ether. A four-day schedule was found to be convenient. Frozen samples were allowed to thaw at room temperature for 48 hours before any work was begun. A portion of the thawed sample was placed into a 100 ml beaker, loosely covered with aluminum foil, and air dried for 24 hours. An air dried sample was thoroughly mixed and 55.0 g weighed onto aluminum foil. The sample was transferred into a 500 ml widemouth Erlenmeyer flask and moistened with 25 ml of 95 percent ethyl alcohol. One hundred ml of Nanograde acetone was added and gently swirled. Samples were covered and allowed to set at room temperature for 24 hours with occasional swirling. The acetone was decanted through Whatman No. 4 filter paper using vacuum suction, transferring a minimum amount of soil. The soil was washed with two small portions of acetone which were decanted through the filter paper. The acetone filtrate was transferred to a 1-liter separatory funnel; 500 ml of distilled water and 100 ml of saturated sodium chloride were added; and these were mixed. One hundred ml of Nanograde petroleum ether was added and shaken vigorously. After separation of the layers, the lower aqueous layer was discarded. The petroleum ether was washed twice by passing 100 ml of distilled water through, discarding the water. The petroleum ether was transferred to a 250 ml beaker and allowed to air evaporate under a hood to approximately 5 ml. Water was removed from the petroleum ether by adding anhydrous sodium sulfate. The solution was transferred with hexane washing into a 25 ml graduated Erlenmeyer flask and diluted to 10 ml with Nanograde hexane. A 1 to 10 dilution was made and injected into a gas chromatograph. Quantitation was made by comparing peak heights of samples to peak heights of standards. Florisil column clean-up was used if necessary.

To determine the dry weight, a separate 2.0 g of air dried soil was weighed into a tared weighing dish and allowed to oven dry at 105° C for 24 hours. Samples were allowed to cool to room temperature in a desiccator before reweighing. The dry weight was used for calculation of concentration.

Table 3. Detection Limits for Organic Pesticides

Compound	*Soil/sediments (ppb)*	*Liquid (µg/l)*
Lindane	4.0	0.05
DDE, DDD, DDT	9.0	0.10
Dieldrin	9.0	0.10
Endrin	9.0	0.10
Kelthane	40.0	0.50
Malathion	40.0	0.50
Parathion	40.0	0.50
Tetradifon	80.0	1.00
Guthion	4000.0	50.00
Captan	40.0	0.50
Folpet	40.0	0.50

LIMNOLOGY OF LAKES AND STREAMS

Cedar Lake is a man-made impoundment designed for multiple purposes which include drinking water, boating, swimming, fishing, and hunting, as currently practiced, and possibly other uses in the future. A fundamental understanding of the physical, chemical, and biological characteristics of the lake waters is essential for maintaining the water quality necessary to sustain these uses. This study not only provided the opportunity to assess certain natural phenomena that occur in an impoundment but also allowed the development of baseline data that will be useful for in-lake investigations in the future.

Physical Characteristics

In Illinois, lakes in excess of 18 feet in depth (Kothandaraman and Evans, 1971) generally stratify as measured by temperature and dissolved oxygen changes. However, where a wide expanse of water is exposed to a brisk prevailing wind, thermal stratification may not occur. This is not the case for Cedar Lake and Little Cedar Lake because they are generally protected by woodland and high bluffs.

Summer thermal stratification occurs in three distinct water layers. The epilimnion, the upper layer, is the mixed and warmest layer. Circulation by the wind keeps its temperature uniform. The middle layer, the metalimnion, is a zone of rapid changes in temperature with at least 1° C change for each 3 feet of depth. Its thickness and depth below the water surface is variable. The lower layer, the hypolimnion, is the coldest layer, and it is isolated from the upper layer.

The thermal and dissolved oxygen (DO) properties of an impoundment are related. Two main sources of DO exist. The atmosphere is the most reliable source whereby the water is 'reaerated' by oxygen absorption and subsequent

molecular diffusion. The other source is oxygen liberated during the process of photosynthesis. The isolation of the hypolimnion during thermal stratification prevents its oxygenation by atmospheric sources, and influencial photosynthetic activity is not apt to occur in that layer in the absence of light.

Where a lake bottom contains organic matter requiring stabilization by aerobic bacteria, any DO in the hypolimnetic waters is usually depleted. In the absence of DO, certain chemical constituents are reintroduced into solution (in a reduced state) from the bottom sediment. These include ammonia, ferrous iron, manganous oxide, phosphorus, heavy metals, and hydrogen sulfide. These constituents will accumulate in the hypolimnion.

For the thermal and oxygen measurements on the lakes, an attempt was made to select sampling sites in the deeper portions of the water bodies. The stations (see figure 3) are as follows:

1 Water intake, Cedar Lake
2 Poplar Camp Creek leg, Cedar Lake
3 Clay Lick Creek leg, Cedar Lake
4 Cedar Creek leg, Cedar Lake
5 Near inlet, Little Cedar Lake
6 Near dam, Little Cedar Creek

Transects of these stations are shown in figures 8 and 9.

In addition to field measurements for temperature and DO, all as part of the effort to define some physical characteristics of the water, Secchi disc measurements for water transparency were made. These observations are summarized here.

Thermal Profiles. Profiles for temperatures recorded in Cedar Lake (station 1) and Little Cedar Lake (station 6) are shown in figures 10 and 11, respectively. During April and May 1976 the water was almost isothermal, varying from about 14 to 18 C for station 1, and from about 16 to 20 C for station 6. The warmer temperatures were recorded, as expected, in the uppermost water layers. In June, thermal stratification became apparent at both stations, especially station 6, and it was very pronounced during July and August. During these months at station 1, the temperature maximum was 29.5 C in the upper 2 feet and the minimum was 16 C in the lower layers. For station 6 the maximum was 28.6° C, and the minimum 16.0 C. Isothermal conditions were regained at both stations in October following 'fall turnover.' When air temperatures cool the surface layer in the fall, the cooler water becomes heavier and moves downward displacing the hypolimnion water and causing a complete recirculation called the fall turnover.

The graph for station 6 on July 29 depicts a classical summer stagnation period in which the epilimnion extending about 6 feet from the water surface is isolated from the hypolimnion by a thermocline from the 6-foot to the 16-foot depth. From that point the hypolimnion extends to the bottom of the lake.

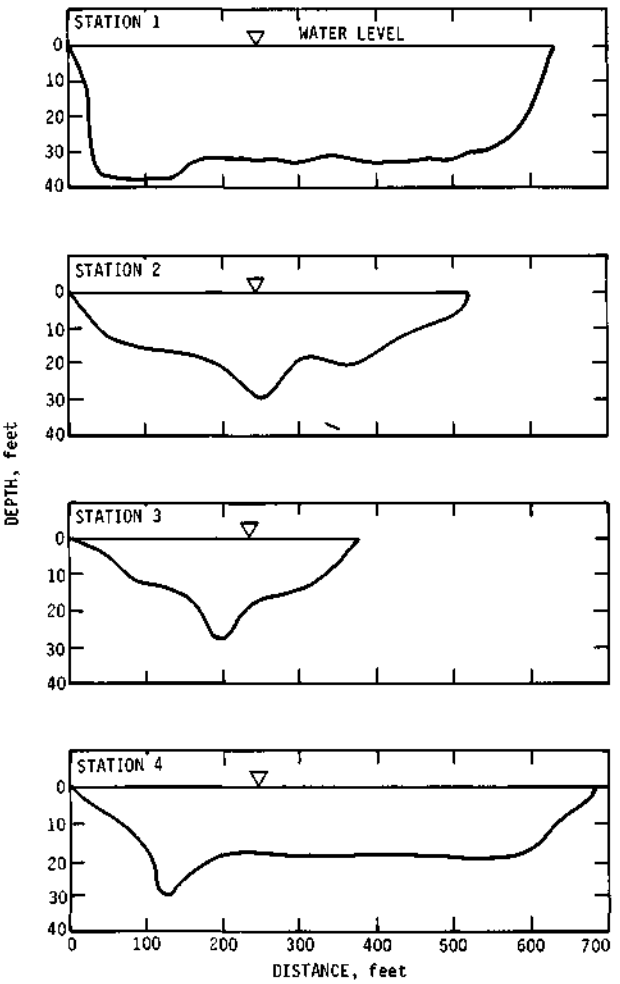

Figure 8. Transects of sampling stations on Cedar Lake

It is during the period of summer stagnation that major water quality changes occur within the lake system. The hypolimnion, by virtue of its stagnation, traps sediment materials and, in the absence of oxygen, causes release of undesirable substances which will be transported to the overlying waters during the fall turnover. The thermal regime for the other lake stations was similar to that for stations 1 and 6.

Dissolved Oxygen Profiles. DO profiles for stations 1 and 6 are shown in figures 12 and 13, respectively. As shown in the graph for April 14, 1976, DO stratification had started before initial field observations. On June 24 at station 6, DO had been depleted in the water column for a depth of 15 feet to the bottom. On the same date at station 1, DO was exhausted from a depth of about 23 feet to the bottom.

In July the severest case observed for each station occurred. In Little Cedar Lake (station 6) on July 29 only that layer of water above an 8-foot water depth contained dissolved oxygen, and in Cedar Lake (station 1) only the up-

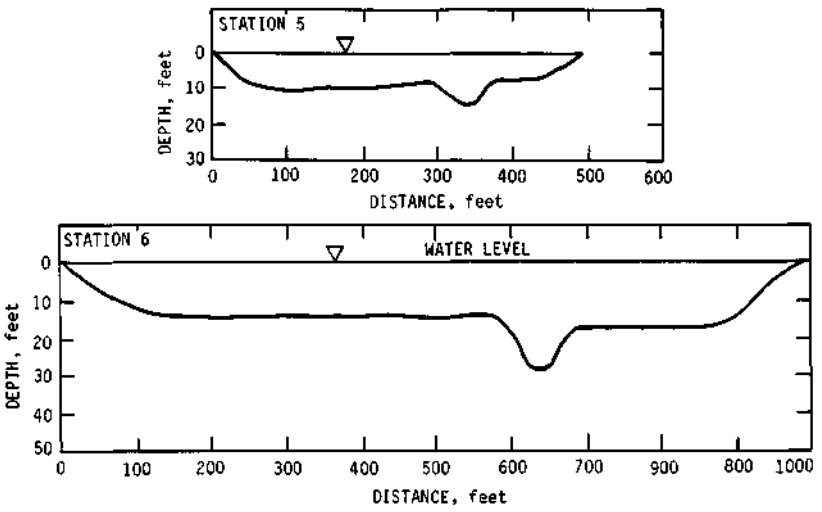

Figure 9. Transects of sampling station on Little Cedar Lake

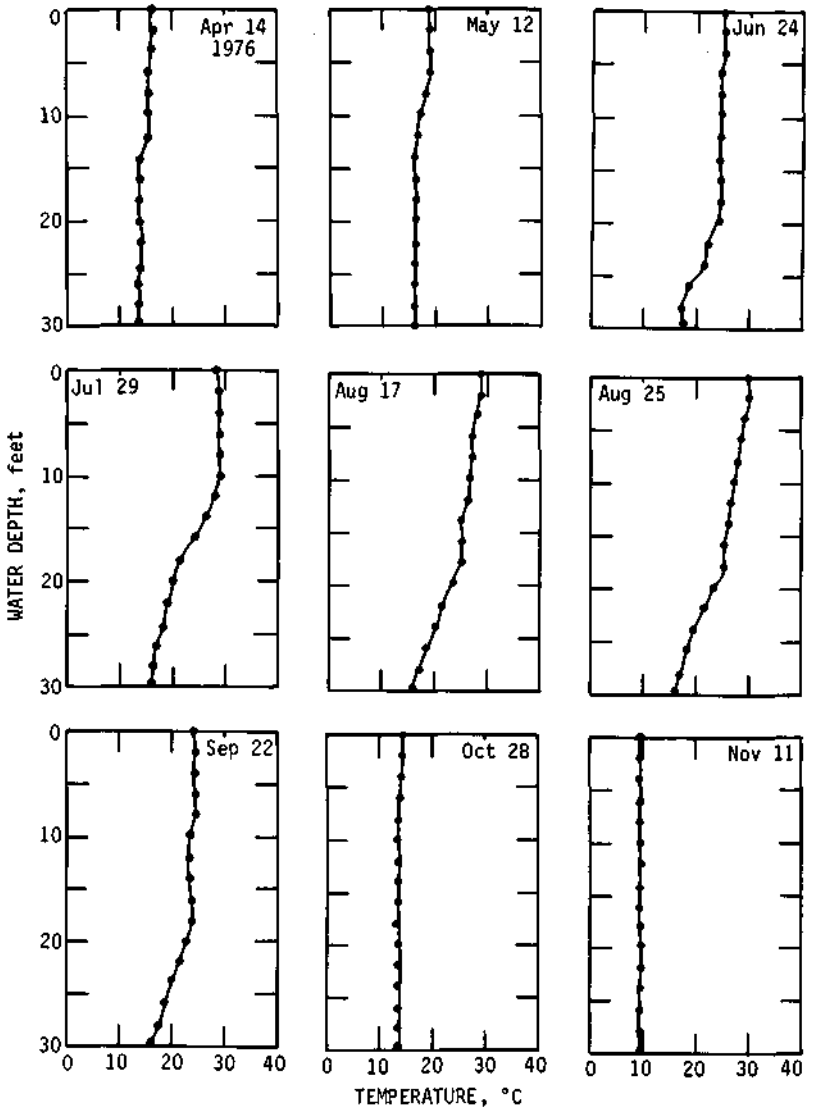

Figure 10. Temperature profiles in Cedar Lake (station 1)

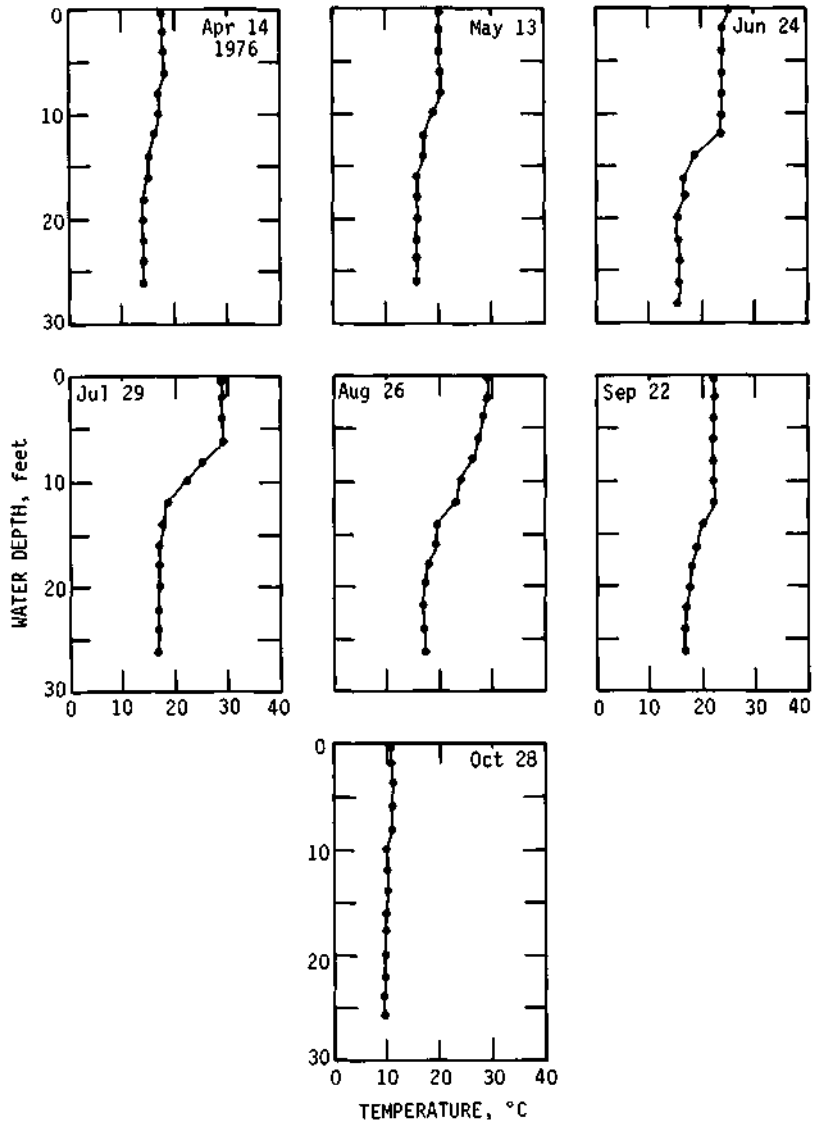

Figure 11. Temperature profiles in Little Cedar Lake (station 6)

per 16 feet of the water contained dissolved oxygen. From July to the end of the field work, conditions improved and there was no DO stratification on October 28.

The concept of DO depletion in hypolimnetic waters is considered substantial evidence that a lake system has aged to the condition of eutrophication — a prelude to extinction. However, it has been found that the hypolimnetic waters of man-made impoundments become without oxygen (anaerobic) within a year of their formation (Kothandaraman and Evans, 1975).

The rate and extent of oxygen depletion in the lake waters at station 6 exceeded that at station 1. This is most likely due to the more enriched lake muds that have accumulated in Little Cedar Lake during the 8 years of its existence.

The impact of dissolved oxygen depletion can best be demonstrated by considering the volume of water being affected. At the time of field observations the elevation of the water surface of Cedar Lake was about 427.5 feet msl. From stage-storage relationships (Stanley, 1968) for Cedar

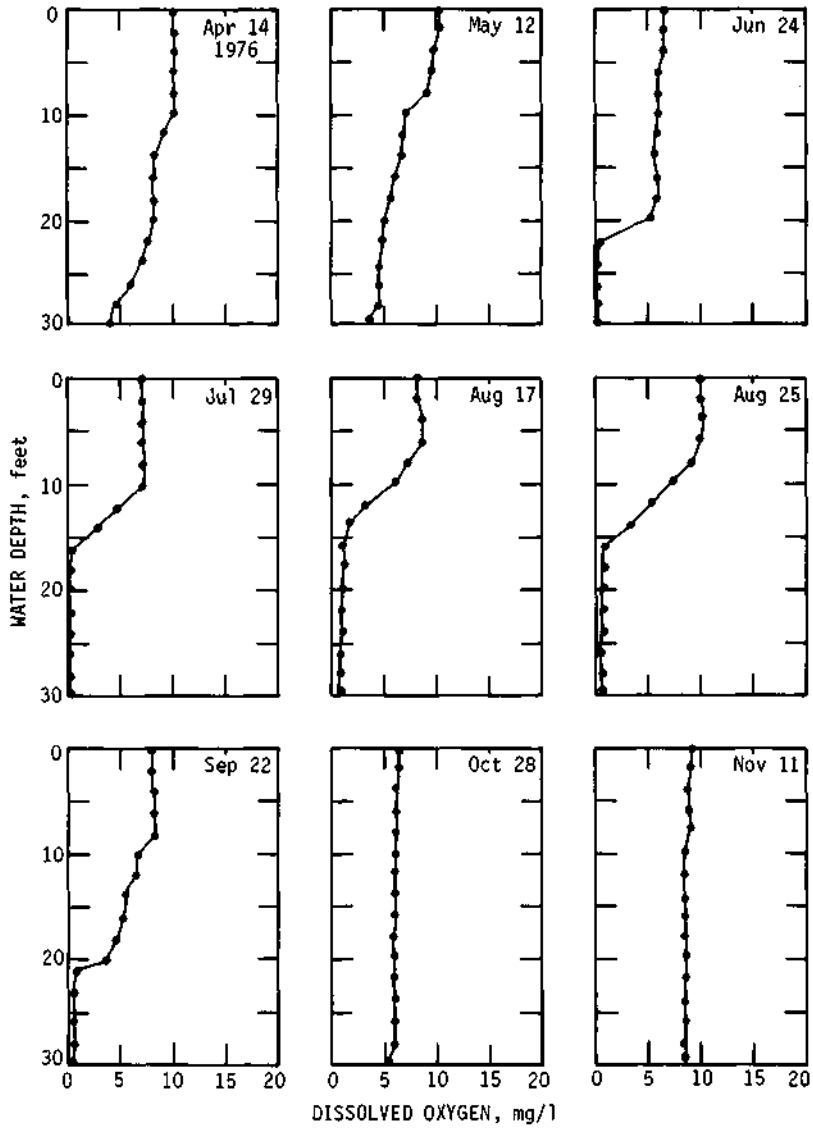

Figure 12. Dissolved oxygen profiles in Cedar Lake (station 1)

Lake, as shown in figure 14, the volume of water within the system was about 35,000 acre-feet. At the 16-foot water depth (elevation 411.5 feet msl) the storage is about 14,250 acre-feet. This is the volume of water without dissolved oxygen on July 29; thus, about 41 percent of the lake water volume was devoid of dissolved oxygen on that date.

Transparency, Secchi disc visibility is a measure of lake water transparency, or its ability to allow light transmission. Though the procedure is not a quantitative measurement of light transmission, it serves as an index by which to compare the waters of the same body of water at different locations and times. Transparency is related to water color and turbidity. For man-made impoundments in Illinois, the reduction in transparency is most often caused by suspended particles of clay and/or silt. Only six to nine observations at the six lake stations were made, which are not sufficient to permit comparison with other Illinois waters but do allow

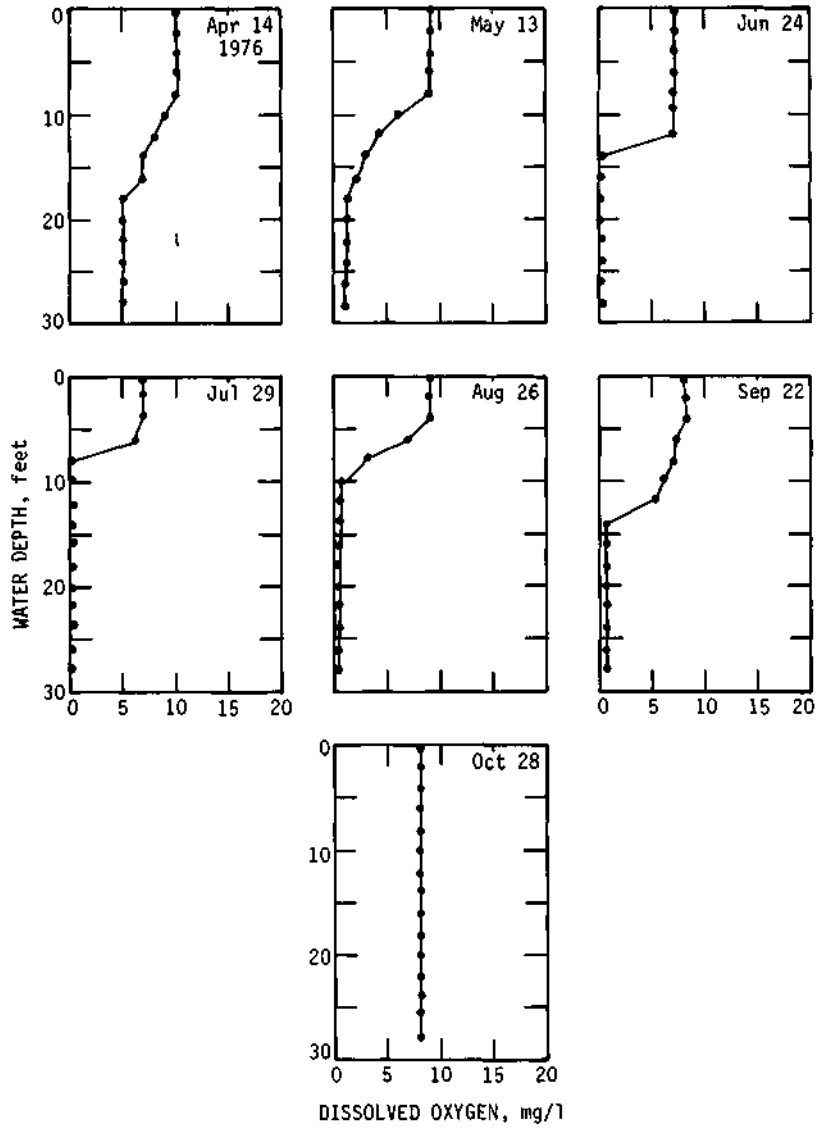

Figure 13. Dissolved oxygen profiles in Little Cedar Lake (station 6)

comparison between the stations. The maximum, minimum, and mean Secchi disc readings are depicted in figure 15. The maximum depth was 54 inches at station 1. The values are reasonable for deeper bodies of water generally sheltered from sustained winds.

Summary. Thermal and dissolved oxygen stratifications do occur in the waters of Cedar Lake and Little Cedar Lake. This is not unusual for man-made impoundments in Illinois. The rate and extent of dissolved oxygen depletion in Little Cedar Lake may be indicative of what to expect in the waters of Cedar Lake 10 years hence. Adequate knowledge of DO and temperature stratification may be useful in selecting water withdrawal depths from Cedar Lake as well as developing fishery management schemes. The transparency of the water suggests that the interference to light transmission is minimal.

Chemical Characteristics

The chemical characteristics of a water body govern not

only the treatment requirements for specified uses but also the magnitude of fluctuations that will occur during seasonal changes. Water samples were obtained on seven occasions from the four Cedar Lake stations and the two Little Cedar Lake stations. As previously described, samples were obtained from the surface, about 3 feet from the bottom, and at the mud-water interface.

Analyses were performed for alkalinity, hardness, ammonia and nitrate nitrogen, total silica, total phosphorus and dissolved orthophosphorus, algal growth potential (AGP), and pH. On one occasion, water samples were also obtained from the major tributary streams to the lakes. The analyses performed were similar to those for the lake waters.

The mean values in the graphs in this section were derived from combining the values for the four sampling locations on Cedar Lake into one mean value for each of the water strata, i.e., surface water, bottom water, and interface. Mean values were similarly developed for Little Cedar Lake.

Alkalinity and Hardness. Alkalinity is a measure of bicarbonate salts in Illinois waters. Hardness is caused principally by calcium and magnesium. Both alkalinity and hardness are governed by the geochemistry of the watershed. Alkalinity provides resistance to pH changes and is therefore the measure of the 'buffering' capacity of a water. The distinction between hard and soft water is relative and arbitrary. The Water Survey (Harmeson and Larson, 1969) classifies hardness in terms of $CaC0_3$ in mg/l as: soft 0-75; fairly soft 75-125; moderately hard 125-250; hard 250-400; and very hard, over 400.

The mean values for alkalinity are shown in figure 16. Values generally ranged from 40 to 65 mg/l as $CaC0_3$. Alkalinity concentrations at the water surface were lower during summer months than in spring or fall. This frequently happens in impounded water and is likely a result of biological activity such as algae production. On a scale of high to low, the waters of the lakes would be classified as low alkalinity waters with concurrent low buffering capacity.

The mean values for hardness are shown in figure 17. Hardness on the average fluctuates from 65 to 85 mg/l as $CaCO_3$ at the surface. Waters close to the bottom increase in hardness during summer stagnation periods as material from the bottom muds become solubilized in the overlying waters. The lake waters are classified as fairly soft.

Nitrogen. Nitrogen is present in water either as dissolved organic nitrogen, or as inorganic nitrogen such as ammonia, nitrate or nitrite, or as elemental nitrogen. In this investigation only the inorganic forms of ammonia-nitrogen (NH_3-N) were examined.

It has been reported (Vollenweider, 1968) that these two forms are used as nutrients by planktonic algae to about the same degree. However, there is evidence (Wang et al., 1973) that during periods of maximum algal growth under laboratory conditions, NH3-N is the form preferred by algae. On the other hand, NH_3-N in the aquatic environment can be very toxic to aquatic organisms, especially fish (Roseboom and Richey, 1977), and it has been documented (Butts et al., 1975) as a significant source of oxygen demand in stream water.

Mean values for NH_3-N concentrations are shown in figure 18. NH_3-N levels in the water at the surface are not significant until after the fall turnover (October). During the period of summer stagnation NH_3-N levels do build up in the bottom waters presumably as a product of bacterial reduction activity in the bottom muds. As the lake waters overturn, NH_3-N is distributed throughout the water column thus elevating the concentration in the upper layer and lowering concentrations, through dilution, in the lower layer. This is a good example of the mechanism of transferring undesirable products of bottom origin to the overlying waters of a lake. Concentrations of NH_3-N were detected as high as 2.54 and 3.86 mg/l in the bottom waters and at the mud-water interface, respectively. Waters at the surface never exceeded 0.45 mg/l.

In addition to $N0_3$-N being considered an important nutrient for algal uptake, it is also a major public health consideration (U.S. Department of Health, Education, and Welfare, 1962). Currently the drinking water standards require that NO_3-N concentrations not exceed 10 mg/l. Nitrate concentration in the lakes did not exceed 0.3 mg/l. This is rather typical of lakes in southern Illinois where NO_3-N concentrations are usually less than 1.0 mg/l. It is not unusual to have mean concentrations of NO_3-N in central Illinois impoundments exceeding 5.0 mg/l (Evans and Schnepper, 1974).

Mean values for NO_3-N are shown in figure 19. Concentration levels in all water strata were fairly constant suggesting that NO_3-N, at the levels detected, are not significantly influenced by dilution or biological activity within the lake systems.

Silica. An abundance of silica in water, along with other nutrients, favors the growth of the plankters, diatoms. Because diatoms contain significant quantities of silica in their skeleton structure, they can be expected to alter the silica concentrations of water bodies. A high degree of inverse relationship between silica and diatom populations in the Illinois River has been reported (Wang and Evans, 1969). Diatom populations constitute about 60 to 90 percent of algal counts in that river.

Silica in Illinois waters is not limiting from the standpoint of algal productivity. The documentation of its concentration levels in Cedar Lake is limited generally to its importance as an indicator of the effect of the lake's bottom on overlying waters. Mean values for silica levels are shown in figure 20. The temporal variations show the accumulation of silica in the lower strata during summer stagnation peri-

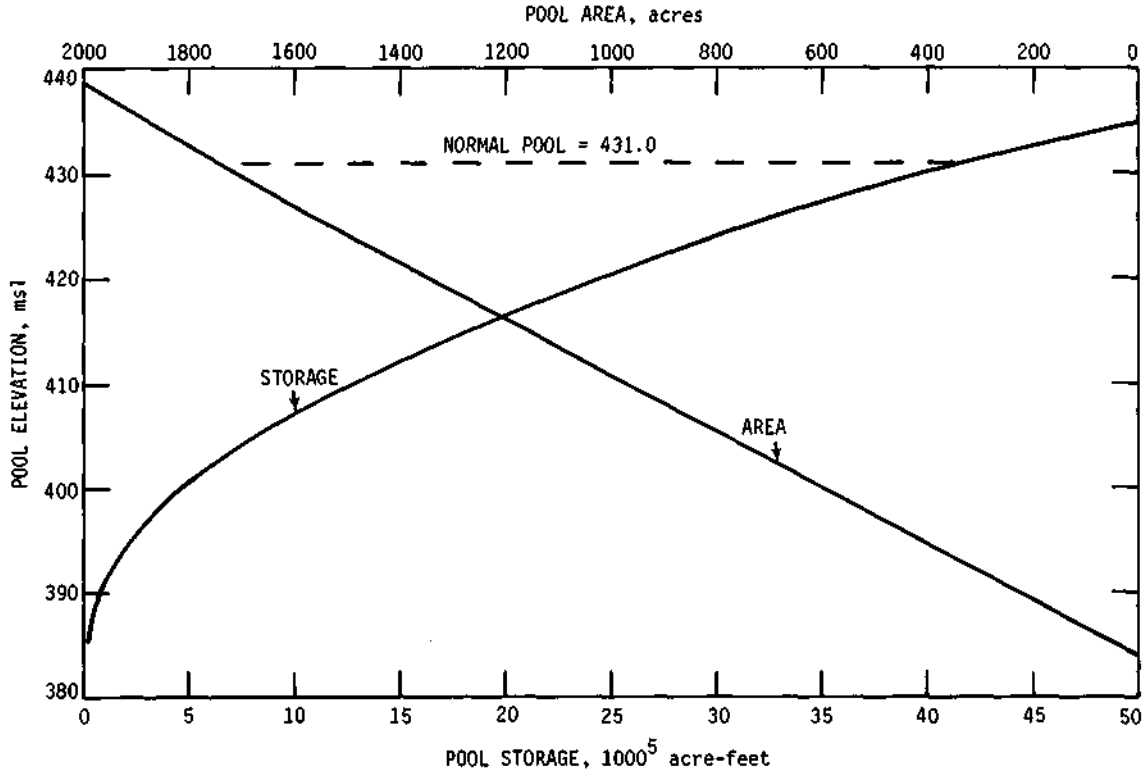

Figure 14. Pool elevation and volume and area relationships for Cedar Lake

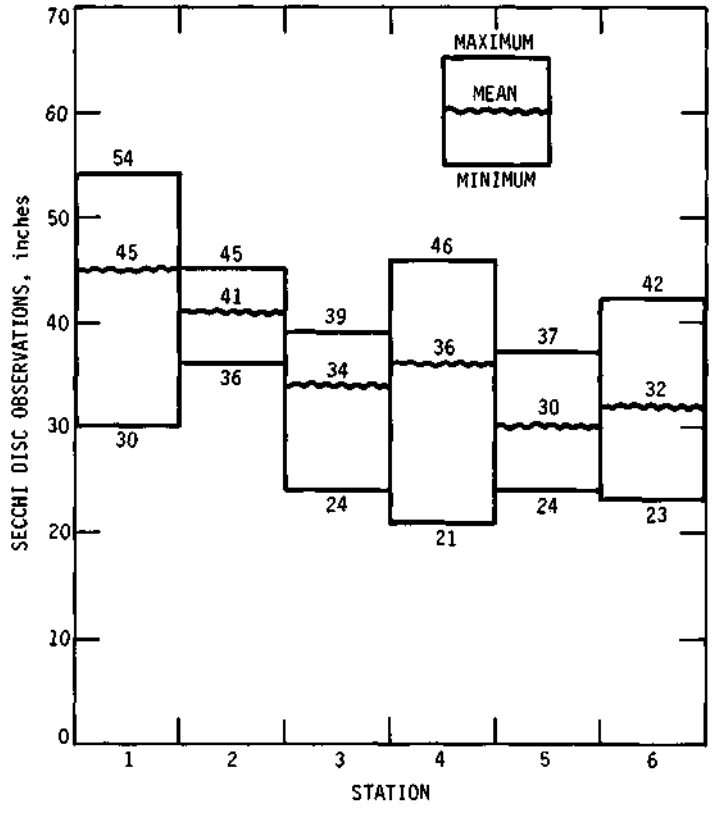

Figure 15. Secchi disc observations, Cedar Lake

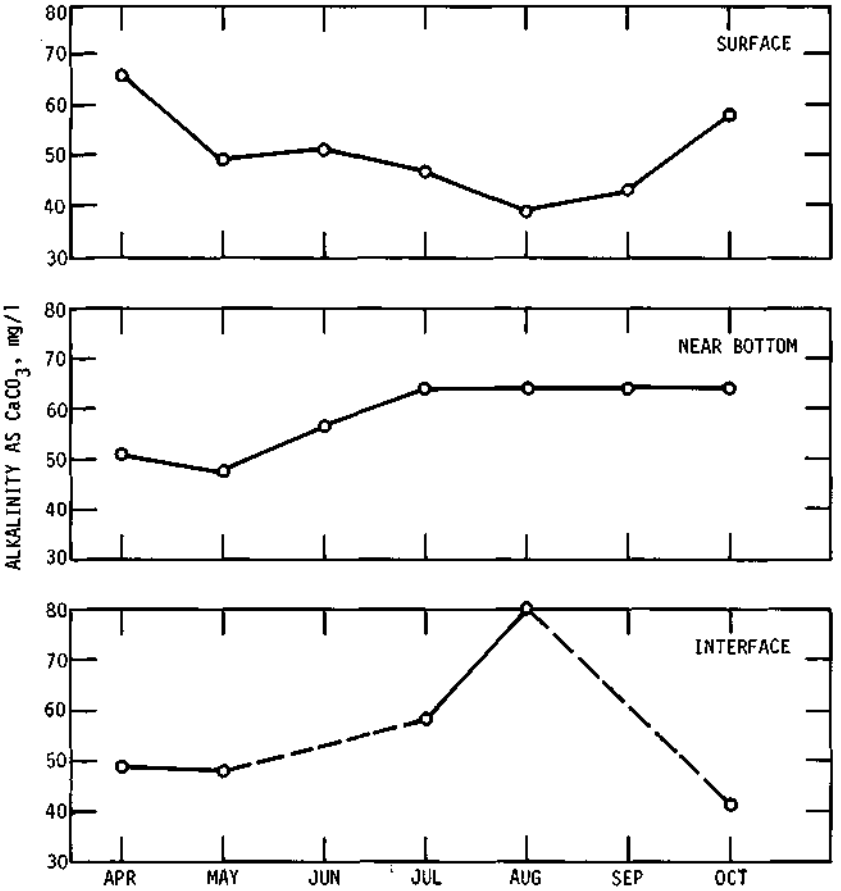

Figure 16. Temporal variations in alkalinity, Cedar Lake

ods. This same type of occurrence was reported (Kothandaraman et al., 1977) for the glacial lake system making up the Fox Chain of Lakes in northeastern Illinois. The overall mean value for the surface water in Cedar Lake is 1.4 mg/1. This is less than that in other lakes in the southern region of the state (Evans et al., 1974). The silica content of Little Cedar Lake is substantially higher than that observed in Cedar Lake, but the reasons for this are not clear-cut.

Total Phosphorus and Dissolved Orthophosphorus. Phosphorus availability is the critical factor governing biological productivity in Illinois waters (Evans and Schnepper, 1974). Total phosphorus includes particulate and dissolved phosphorus. The various fractions and forms of phosphorus in Illinois waters have been discussed by Sullivan and Hullinger (1969). For this study, only total phosphorus and dissolved orthophosphorus were examined. Dissolved orthophosphorus is the form considered more readily available for biological uptake.

The mean values for total phosphorus and dissolved orthophosphorus are shown in figures 21 and 22, respectively. At the water surface the mean total phosphorus concentrations were less than 0.05 mg/1. This is substantially less than that observed on the highly eutrophic lakes of the Fox Chain (Kothandaraman et al., 1977) where the mean values were frequently in excess of 0.20 mg/1 at the water surface. In May, total phosphorus was not detected in Cedar Lake. However, during summer stagnation mean values of 0.12, 0.18, 0.58, and 0.25 mg/1 were observed in the bottom waters during June, July, August, and September, respectively. At the interface, corresponding values for July, August, and October were 0.44, 1.23, and 0.35 mg/1, respectively. These relatively high values in the lower strata

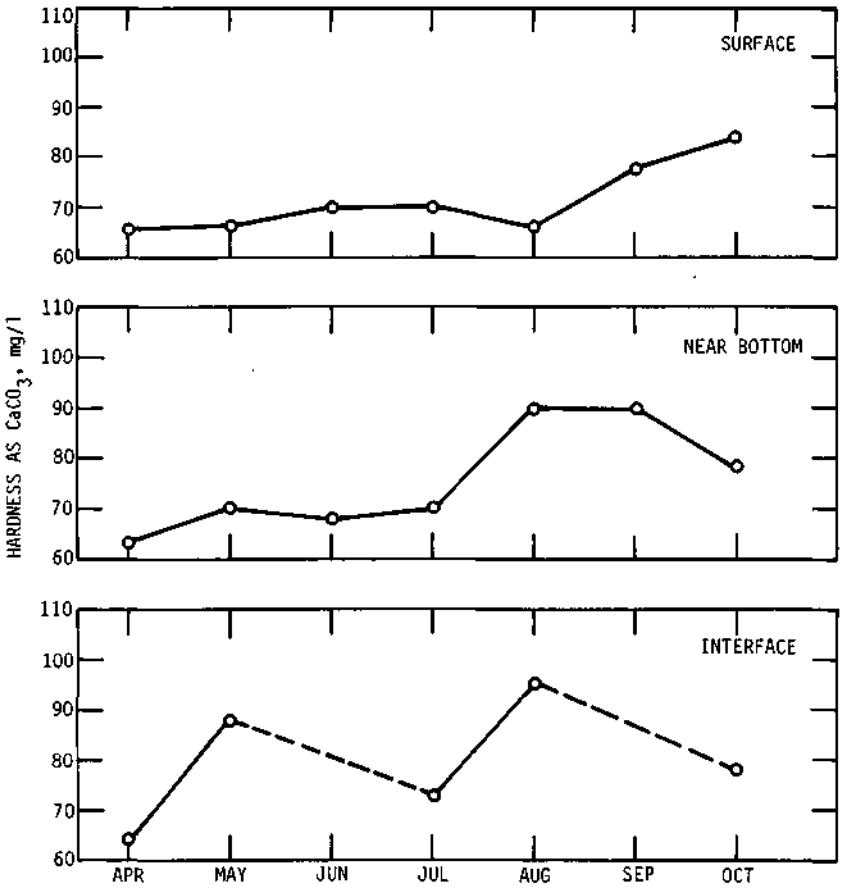

Figure 17. Temporal variations in hardness, Cedar Lake

emphasize the importance of the bottom muds as a source of nutrients for the lake system.

Dissolved orthophosphorus was detected only during the months of April and July at the water surface. Concentrations ranged from 0.00 to 0.01 mg/1 at the surface. They reached a maximum mean in the bottom waters of 0.12 mg/1 in June and a maximum of 0.09 mg/1 in August at the mud-water interface. It is important to note that figures 21 and 22 have different ordinate scale values.

Algal Growth Potential. As previously discussed, the algal growth potential of a water body provides a relative estimate for comparing the capability of various waters or the same water during different seasons to sustain algal growth. Mean values for AGP are shown in figure 23. The AGP of the surface water ranged from 4 to 25 mg/1 with a mean of 10 mg/1. The mean values for the bottom waters and interface were 12 and 11 mg/1, respectively. These values are substantially less than those observed in 16 other water supply impoundments (Evans et al., 1974) where the mean AGP ranged from 10 to 65 mg/1.

Evans and Schnepper (1974) concluded that when mean AGP values are equal to or less than 20 mg/1, naturally occurring algal densities of nuisance proportion are unlikely. Mean AGP values greater than 20 but less than 40 mg/1 indicate that nuisance algal densities are likely to occur but can be controlled by chemical means. On this basis, coupled with observed nutrient concentrations, it is unlikely that nuisance algal blooms will occur under present conditions in Cedar Lake.

pH. The pH of a natural water is commonly considered to be a measure of its acid or alkaline nature. A pH of 7 is neutral; that below is considered on the acid side and that

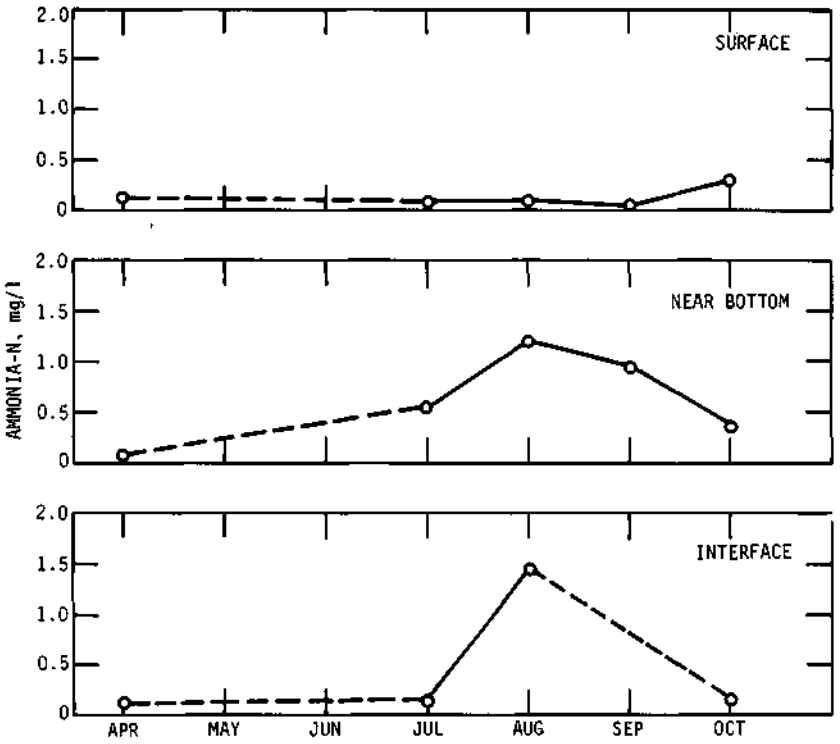

Figure 18. Temporal variations in ammonia-N, Cedar Lake

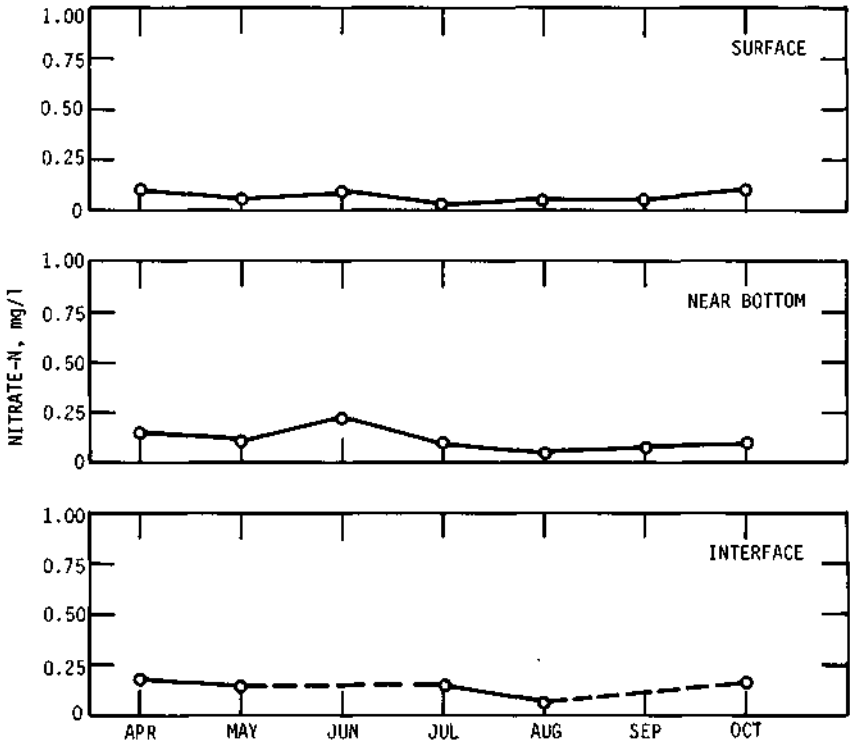

Figure 19. Temporal variations in nitrate-N, Cedar Lake

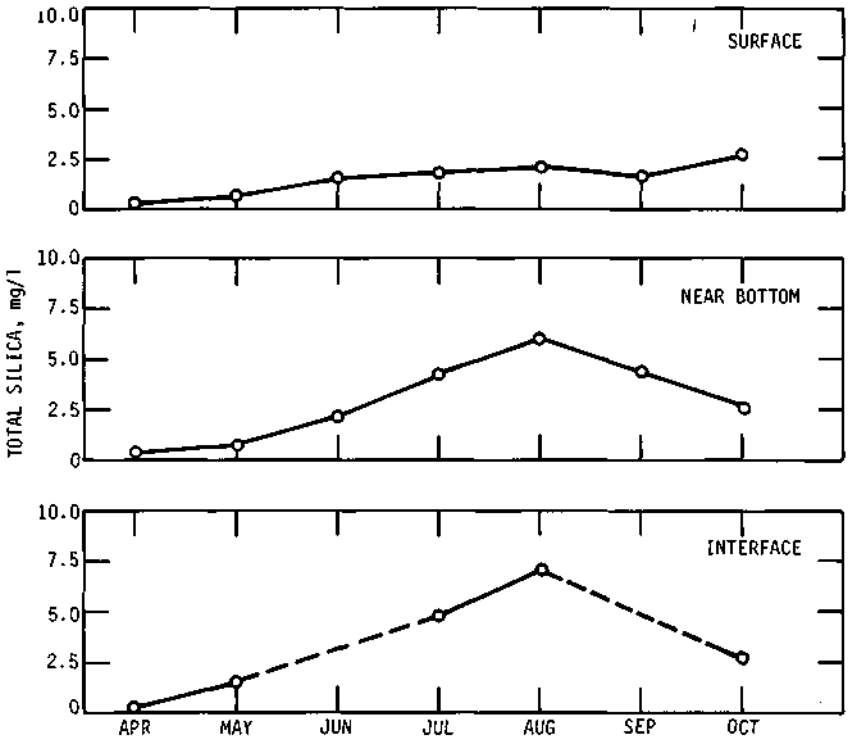

Figure 20. Temporal variations in total silica, Cedar Lake

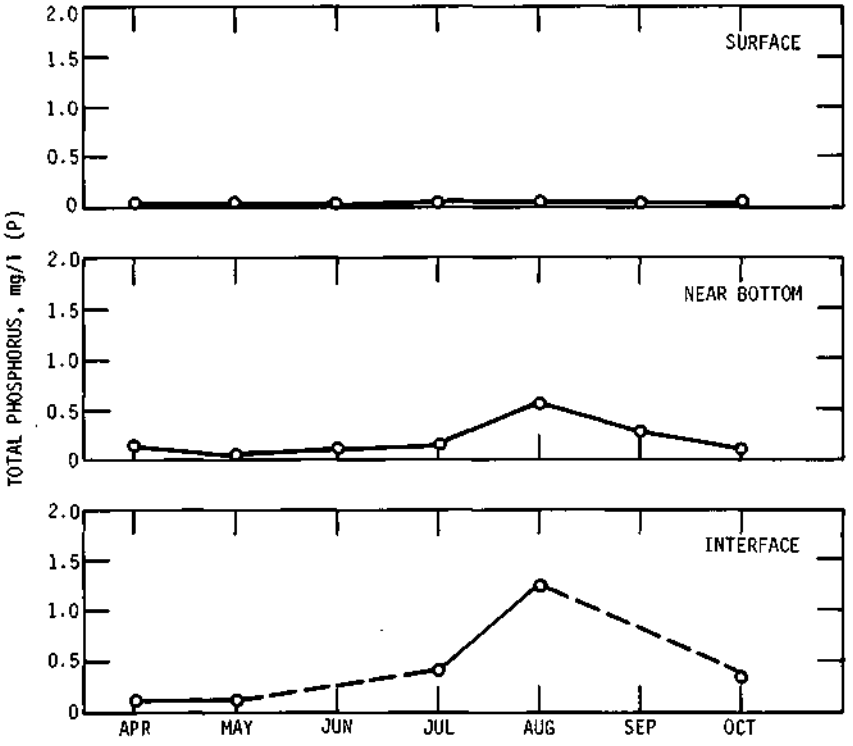

Figure 21. Temporal variations in total phosphorus, Cedar Lake

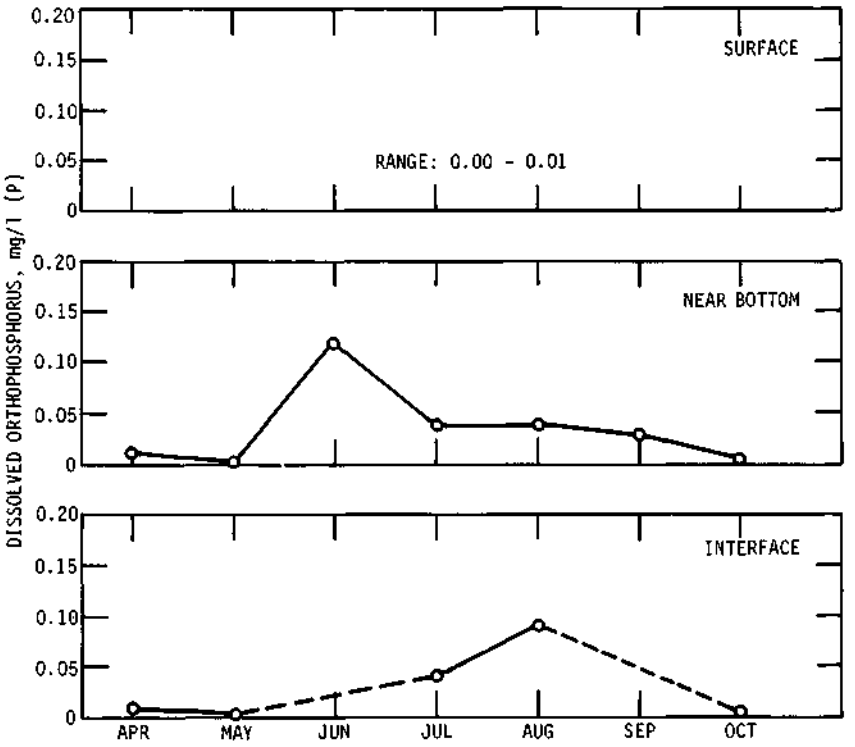

Figure 22. Temporal variations in dissolved orthophosphorus, Cedar Lake

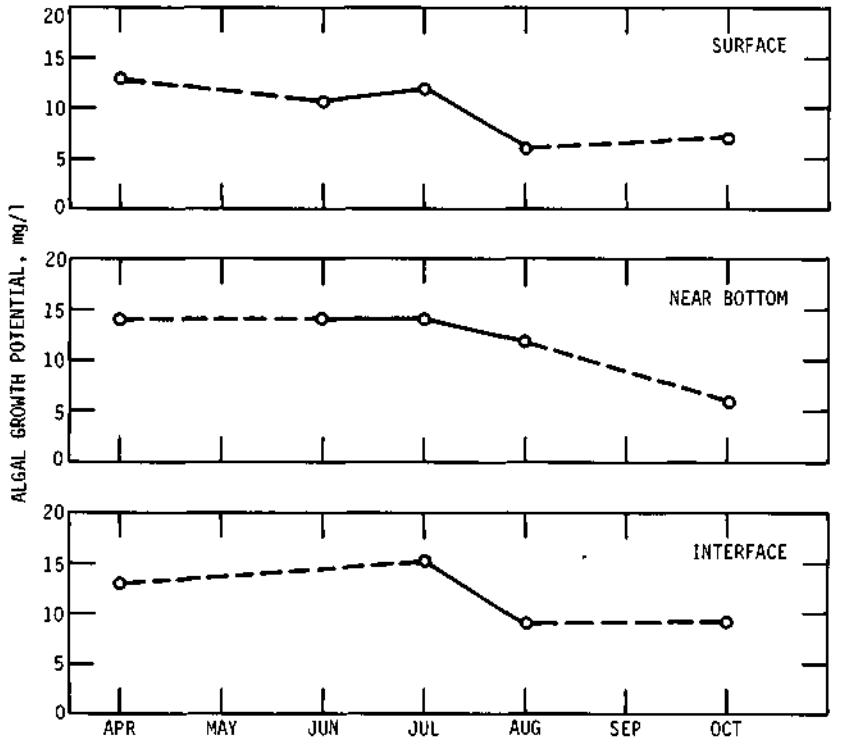

Figure 23. Temporal variations in algal growth potential, Cedar Lake

Table 4. Water Chemistry of Major Tributaries to Cedar Lake

(Concentrations in mg/l)

	pH	*Alkalinity*	*Hardness*	*NO_3-N*	*Total SiO_2*	*Total phosphorus*	*Dissolved phosphorus*	*Algal growth potential*
West Cedar Creek	7.91	80	82	0.59	12.5	0.31	0.08	4
East Cedar Creek	7.67	60	68	0.32	8.6	0.31	0.00	34
Clay Lick Creek	7.88	72	88	2.09	7.4	0.38	0.12	101
Mill Creek	7.83	64	88	0.60	7.7	0.16	0.00	4
Poplar Camp Creek	7.91	76	82	0.99	8.7	0.26	0.01	3

Table 5. Comparison of Concentrations of Some Chemical Constituents in Nearby Impoundments*

(Concentrations in mg/l)

	Alkalinity	*Hardness*	*NO_3-N*	*NH_3-N*	*Total SiO_2*	*Total phosphorus*	*Dissolved phosphorus*	*Algal growth potential*
Alto Pass Reservoir	49	65	0.51	0.21	7.2	0.06	0.03	26
Ashley Reservoir	50	87	0.68	0.18	3.4	0.03	0.01	18
Centralia Reservoir	44	95	0.47	0.09	4.2	0.03	0.01	14
Marion Reservoir	80	143	0.17	0.06	2.7	0.03	0.02	8
Waterloo Reservoir	101	123	0.37	0.09	2.1	0.02	0.01	14
Cedar Lake	50	71	0.06	0.11	1.4	0.03	<0.01	10

**Mean values were based on 26 to 28 samples per location*

Table 6. Means of Some Chemical Constituents in Cedar Lake at Three Depths

(Concentrations in mg/l)

	Alkalinity	*Hardness*	*NO_3-N*	*NH_3-N*	*Total SiO_2*	*Total phosphorus*	*Dissolved phosphorus*	*Algal growth potential*
Surface	50	71	0.06	0.11	1.44	0.03	0.01	10
Near bottom	60	76	0.12	0.62	2.97	0.18	0.03	12
Interface	56	79	0.13	0.58	3.28	0.44	0.02	10

above is considered on the alkaline side. The pH of a natural water will fluctuate with the photosynthetic activity of phytoplankton. The waters of Cedar Lake showed pH variations between 7.0 and 8.0 which is considered normal.

Stream Waters. Water samples were obtained from the major tributaries to Cedar Lake on one occasion for chemical analyses. The results are shown in table 4. Though a worthwhile evaluation of such limited data is not possible, it should be noted that the concentrations of the chemical constituents in the streams are generally in excess of those same constituents observed in the lake water. This relationship is not unlike that observed for other tributary-lake systems, and demonstrates the modifying influence of lake waters.

Summary, The concentrations of measured chemical constituents in Cedar Lake are similar to those examined in other man-made impoundments in southern Illinois. Table 5 shows that silica and nitrate-nitrogen concentrations are less than in other regional impoundments, but other constituent concentrations are about the same.

The principal changes within the lake system are limited to ammonia-nitrogen, silica, and total phosphorus. As shown in table 6, there is considerable difference in concentrations between the upper and lower water strata among these elements. The variations occur during summer stagnation and are the result of chemical and biological activities within the lake muds during the absence of dissolved oxygen.

Biological Characteristics

The study of a lake system also requires an examination of some of the aquatic organisms being supported within its waters. Therefore, an effort was made to develop some information on the algae and benthic organisms existing in the waters of Cedar Lake and Little Cedar Lake.

Algae are part of the phytoplankton population, those organisms that are free-floating in the water. The benthic organisms (benthos), although capable of migrating within a water column, generally reside in the lake bottom muds. Both types, algae and benthos, may alter the physical and chemical quality of a water body, but their principal status is that of providing a food supply for higher aquatic organisms such as crustaceans and fishes.

Samples for algae identification and enumeration were obtained on six occasions from the surface waters of Cedar Lake and Little Cedar Lake. These were obtained during the months of April through October, and the dates of collection were generally the same as for the water samples.

Benthic organisms were collected on four occasions (April, May, July, and October) from Cedar Lake, on three occasions (April, July, and October) from Little Cedar Lake, and on one occasion (August) from the Alto Pass Reservoir.

The number and types of organisms collected and their significance to the water quality of their respective lakes are discussed here.

Algae Types. Algae are classified in part according to their color. The blue-green algae are so named because they reflect a blue to dark green tint. In massive numbers (a bloom) they create a 'green paint slick' upon the water surface. Most blue-green algae grow in nonfilamentous colonies. They are widely distributed but are more frequently found in lakes and ponds than in the running waters of streams. When found in flowing streams in Illinois their source is likely to be an upstream impoundment.

The green algae usually contain one major group of pigments, the chlorophylls. They may be either free-floating or attached. If numerous, they display a green cast to the water.

In several divisions of algae, including some greens, there are species that are unicellular and equipped with flagellum, a whiplike organ that permits these algae to be mobile. These algae are flagellates. The cells range from spherical to ovoid depending on the species.

Somewhat intermediate between the blue-green and green algae is a group known as diatoms. They are the most numerous in Illinois waters. Diatoms are characterized by the presence of silica in their cell walls, and by the presence of green, yellow, and brown pigment associated with the chlorophylls, depending on the stage in their life cycle. They vary in color from brown to green. Generally the cell is oblong to circular, although there can be a variation in shape.

Blue-green algae were not found in Cedar Lake, but they were detected on one occasion at each of the two stations on Little Cedar Lake. Green algae, flagellates, and diatoms were detected at all lake stations.

Algae can impair the recreational use of a lake, create taste and odors in drinking water, and cause the clogging of filters in water treatment plants. They can become so numerous as to impose a significant dissolved oxygen demand in water. Although attempts have been made to 'tag' different genera or species of algae as causing various types of water quality problems, e.g., taste and odor, filter clogging, etc., the arbitrary acceptance of such classifications without considering corollary environmental conditions is questionable.

Algae Composition. During the collection period, 27 algal genera were recovered from 24 samples on Cedar Lake, and 24 algal genera were recovered from 10 samples on Little Cedar Lake. The genera, collectively, included 1 blue-green algae, 8 green algae, 5 flagellates, and 16 diatoms. For comparison with other lakes in the region, the Water Survey has had experience only with Dolan Lake in Hamilton County (Butts et al., 1976). In that lake, 34 genera were recovered from 12 samples obtained at thesurface, mid-depth, and near the bottom on four occasions; the types and genera of the algae were similar to that obtained during this study. In contrast, from collections made during a 5-month period on the eutrophic waters of the Fox Chain of Lakes, 41 algal genera were recovered from 414 samples. Four blue-green genera were among the collections (Kothandaraman et al., 1977).

The most frequently occurring green algae in Cedar Lake was *Actinastrum sp.; Tracbelomonas sp.* was the most frequently occurring flagellate; and the diatoms *Synedra sp., Melosira sp.,* and *Navicula sp.* predominated for their type occurring about 46, 33, and 33 percent of the time, respectively. These algae are commonly considered clean water varieties. The number of genera occurring per station varied from 16 to 19 genera.

Algae Density. Algae density is expressed as cell counts per milliliter (cts/ml). Algal counts ranged from a low of 77 at station 3 to a high of 516 cts/ml at station 1. These are low densities. On the Fox Chain of Lakes, algae concentrations were as high as 14,000 cts/ml. Generally a density of 500 cts/ml is considered an algal bloom, but the nuisance effect is related more to the type of algae than to the number of algae. A count of 500 diatoms is insignificant and visually non-detectable; a count of 500 blue-greens would be noticeable and, from a recreational standpoint, alarming. The geometric mean did not exceed 235 cts/ml at any lake station.

From the standpoint of algal densities, *Actinastrum sp.* and *Crucigenia sp.* made up 50 percent of the green algae population in Cedar Lake. *Trachelomonas sp.* accounted for

68 percent of the flagellate population, and the four diatoms *Navicula sp., Synedra sp., Melosira sp.,* and *Nitzschia sp.* made up 53 percent of the diatoms. Green algae predominated during April but thereafter diatoms were the predominant type. In a lake system not plagued with algal problems diatoms usually predominate.

Benthic Organisms. The benthic organisms (benthos) found can be characterized as pollution tolerant. The only three taxa recovered were Chironomidae, *Chaoborus,* and Tubificidae.

The Chironomidae (midge fly larvae) remain in the bottom sediments until just before the emergence of the adult midge fly. They have a hemoglobin-like blood pigment and special gills to extract sufficient oxygen from nearly oxygen depleted water. The *Chaoborus* has air sacs which permit it to visit the surface waters to feed and renew its oxygen supplies at night. During the day, this organism burrows in the bottom sediments and may survive there in great numbers, even in the absence of oxygen.

The Tubificidae (aquatic worms), in contrast to the other benthos found, are in the adult stage. Many species occur in natural water. Some are tolerant and others are intolerant to pollution. Their lifetime habitat is confined to the bottom muds and they try to burrow partially with parts of their body extending into the mud-water interface. They prefer an organically enriched bottom mud.

Community structure can be used to determine the health or balance of a given habitat, in this case the bottoms of Cedar Lake, Little Cedar Lake, and the Alto Pass Reservoir. A *diverse* community, in terms of benthos, has many species with relatively few individuals in each one. A *simplified* community, indicating harsh environmental conditions, has a few species with large populations in each one. The community structure of the lakes in this study is characteristic of the *simplified* community.

The type and number of benthos collected from each station are shown in table 7. In Cedar Lake, the Chironomidae and *Chaoborus* predominate. There is no particular pattern apparent in terms of dominance among the two taxa. In Little Cedar Lake and Alto Pass Reservoir *Chaoborus* is clearly the dominant taxon. The fact that this benthos has the ability to obtain oxygen by migration at the water surface and Chironomidae do not, suggests that dissolved oxygen conditions in the lower water strata of Little Cedar Lake and Alto Pass Reservoir are more critical than those thus far experienced in Cedar Lake. A very harsh environment does exist in these two impoundments as shown by the dissolved oxygen stratification measurements. The differing population densities (number per square meter) for these two communities demonstrate the value of assessing the benthos for deriving estimates of environmental conditions in a lake system.

Table 7. Abundance of Benthic Organisms

(In number per square meter)

Station	*Date*	Chironomidae	*Chaoborus*	Tubificidae
Cedar Lake				
1	4/14	536	174	44
	5/12	179	1,025	25
	7/29	0	130	0
	10/28	348	493	116
2	4/14	464	232	29
	5/12	556	43	25
	7/29	44	493	0
	10/28	1,420	1,304	0
3	4/14	362	304	0
	5/12	543	74	0
	7/29	0	261	0
	10/28	1,783	739	58
4	4/14	565	971	44
	5/12	1,136	210	37
	7/29	0	406	0
	10/28	449	4,493	0
Little Cedar Lake				
5	4/14	942	7,913	0
	7/29	15	710	0
	10/28	2,102	16,406	0
6	4/14	1,058	2,536	0
	7/29	0	739	0
	10/28	681	16,058	0
Alto Pass Reservoir				
7	8/12	0	5,754	0
	8/12	0	4,551	0

The Tubificidae populations are relatively sparse. This may be due to the lack of organically enriched mud bottoms rather than dissolved oxygen conditions.

Summary. Twenty-seven algal genera were obtained from 24 samples on Cedar Lake. *Actinastrum sp.* was the most frequently occurring green algae. The flagellate *Trachelomonas sp.* was the most frequently occurring algae of its type and the diatoms *Synedra sp., Navicula sp.,* and *Melosira sp.* occurred most often among the silica-structural algae.

Algal densities were low, ranging from 77 to 516 cts/ml. Diatoms usually were the most prevalent algal type, occasionally accounting for 50 percent or more of the population. Nuisance blue-greens were observed only on one occasion and that was on Little Cedar Lake.

From the algal data assembled and the visual observations, the water of Cedar Lake does not support an algal population in terms of either composition or density that interferes with the planned uses of the lake.

PESTICIDES

The pesticides of interest to this study are those that have been and are being used by commercial fruit growers for the control of pests in apple and peach orchards in Illinois. The land acreage supporting apple and peach orchards in Illinois is not vast. In 1968, there were about 560 orchards on 14,270 acres, and in 1975 there were about 417 orchards on 11,400 acres (Illinois Cooperative Crop Report Service, 1968, 1976). This indicates an annual loss of about 2.9 percent during an 8 year period. During this period the reported losses in apple and peach trees were 10 and 23 percent, respectively. However, Mowry (1977) estimates that apple and peach orchard acreage currently occupies about 15,500 acres of Illinois cropland.

Although orchard acreage is not great, the use of pesticides is intensive. Commercial fruit growers in Illinois use an average of 11 to 14 different chemicals for pest control annually (Transactions of the Illinois State Horticulture Society and the Illinois Fruit Council, 1963-1974). Applications of the pesticides are made an average of about 14 to 15 times a year. These pesticides consist of insecticides, miticides, fungicides, bacteriocides, rodentocides, and herbicides. In addition to small quantities of fertilizer, chemicals are also applied to regulate tree growth and to influence fruit growth. According to Mowry (1977):

> "... a season-long pesticide schedule for 1 acre of apple orchard (generously calculating 19 sprays of 400 gal. per acre for each spray application) would deposit 139 lbs. of pesticide (actual ingredients) per acre of soil (63 lbs. of insecticide, 72 lbs. of fungicide, 4 lbs. of herbicide), resulting in a pesticide depth of .0005 in. when spread evenly over an acre of soil.
>
> An acre of peach orchard would receive 245 lbs. of pesticide (actual ingredients) per acre of soil (67 lbs. of pesticide, 174 lbs. fungicide, 4 lbs. herbicide) resulting in a pesticide depth of .0009 in."
>
> [Note: Mowry, for 'hypothetical comparisons,' points out that the values developed for accumulation purposes disregard continuous degradation of each pesticide application during intervals between applications.]

The rates at which pesticides and their by-products degrade under natural conditions are the first consideration in examining their effect on the aquatic environment. Only a limited number of field studies have been reported. The pathway for degradation depends upon temperature, oxygen concentration, and the presence of a viable microbial population. The rate is dependent upon the characteristics of the pesticide. For this report the general characteristics were limited to inorganic and organic types.

The inorganic pesticides most often leave residues that are virtually non-degradable. These residues are often potentially toxic and are represented here by such metals as arsenic (As), lead (Pb), copper (Cu), zinc (Zn), and mercury (Hg). These inorganic pesticide residues (classified frequently as heavy metals or trace elements) will usually be translocated by soil movement in contrast to leaching.

The organic pesticides consist of chlorinated hydrocarbons, organophosphates, and carbamates. The chlorinated hydrocarbons include DDT and its degraded products DDD and DDE. Among these synthetic organic compounds, ranked in order of increasing persistence under anaerobic conditions, are: lindane, heptachlor, endrin, DDT, aldrin, heptachlor expoxide, and dieldrin (Hill and McCarthy, 1967). These compounds are known for their longevity (periods longer than 1 year) and have a strong tendency to sorb onto soil sediment and algal particles within natural waters (Cleaning Our Environment, 1969). They are bioaccumulative with their residues becoming magnified with each successive step in the food chain. The degradation of chlorinated hydrocarbons generally occurs more rapidly under anaerobic conditions than in the presence of dissolved oxygen. Their application for agricultural use has been prohibited nationally. However, the persistence of these pesticides suggests that they should not be disregarded in any study of the effect of pesticide applications, past or present.

Organophosphates have replaced chlorinated hydrocarbons for pesticide use in recent years. Among these compounds are malathion, guthion, parathion, and imidan. The organophosphates are considered non-persistent because their residues last only a few months or weeks under normal application rates. They hydrolyze quickly and are readily susceptible to degradation by bacterial activity. No detailed field studies regarding the presence, much less the degradation rates, of the compounds exist.

Another non-persistent group of organic pesticides is carbamates, a comparatively new class of pesticides that contain neither chlorine nor phosphorus. Among them are carbaryl, zineb, dikar, and ferman. As in the case of organophosphates, there is a lack of field studies designed to determine the degradation by-products of carbamates.

The major pesticides used by commercial fruit growers in southern Illinois, in terms of insecticides, miticides, and fungicides for a 12-year period, are shown in table 8. The listing was compiled from a review of the Transactions of the Illinois State Horticulture Society and the Illinois Fruit Council for the years 1963 through 1974. The number of years each chemical was used during the 12-year period is noted. A classification was made where possible. The estimates of bioaccumulativity and biodegradability were obtained from a review of the Cooperative Extension Service (1975) publication regarding a pesticide applicator guide.

With the exception of lead arsenate, the principal insecticides used were organophosphates. One organochlorine substance, Kelthane, was used for mite control and carbamates were limited to fungus control.

Table 8. Major Pesticides Used, 1963-1974

	Years used	Class	Bioaccumulativity	Biodegradability
Insecticides				
1 Guthion	11	Organophosphate	None	High
2 Lead arsenate	7	Inorganic	High	Low
3 Malathion	5	Organophosphate	None	High
4 Parathion	4	Organophosphate	None	High
5 Imidan	4	Organophosphate	None	High
Miticides				
1 Kelthane	3	Organochlorine	High	Low
2 Tedion (Tetradifon)	2			
3 Morocide (Binapacryl)	3			
4 Sevin	2	Carbamate	None	High
Fungicides				
1 Zineb	9	Carbamate	None	High
2 Cyprex (Dodine)	8			
3 Captan (Orthocide)	6	Phthalimide		
4 Dikar	5	Carbamate	None	High
5 Streptomycin	4			
6 Karathane (Dinocup)	4			
7 Niacide	3		None	High
8 Polyram	3	Carbamate	None	High
9 Sulfur	3			

During this study samples were collected from the soils of 10 orchards for the examination of inorganic and organic pesticide residues. For each orchard sampled, an attempt was made to also sample a row crop, pasture, and woodland in its vicinity. The series of samples, representative of four land uses, were assigned location numbers 1 through 10. In addition to soil samples, limited examinations for inorganic and organic pesticide concentrations were also made on the suspended solids and alluvial deposits of the major streams in the watershed, as well as the bottom muds, interstitial waters of the muds, and the water-bottom interface of the lakes.

Inorganic Pesticides

The residues of inorganic pesticides likely to occur in orchard soils include arsenic, lead, copper, zinc, and mercury. In addition to performing analyses for these trace elements in the silt loam of the watershed, examinations were also made for iron (Fe) content. The importance of particle size on the concentration of trace elements has been reported by Rickert et al. (1977). As shown in figure 24, they found that the concentrations of copper, lead, and zinc generally increased as the particle size decreased. Because of this relationship, only that fraction of the soil that passed through a 63-micron opening was examined for inorganic pesticide content. Sampling of soils was performed before significant spraying operations commenced.

Heavy Metals in Soil

The initial plan of sampling visualized a corresponding soil sample from a row crop, pasture, and woodland in the vicinity of each of the 10 orchard locations selected for sampling. However, at some orchard locations not all of the other three land uses were practiced. This led to the examination of soils from 10 orchards, 6 row crops, 10 pastures, and 8 woodlands. In other words, some of the orchard locations did not have row crops and woodlands in their vicinity. The purpose of examining soil samples from land other than that used for orchards was to gather data that would be useful for background purposes. Adequate information on the natural or background concentrations of arsenic, lead, copper, zinc, mercury, and iron in neighboring soils was essential if changes in the concentration of these elements in orchard soils were to be detected.

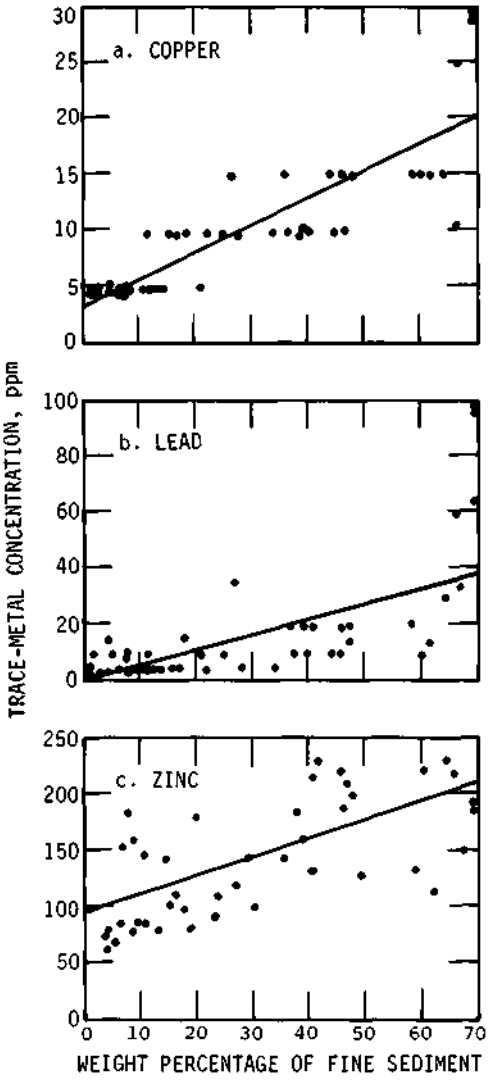

Figure 24. Trace elements concentration versus particle size

To determine probable background concentrations, three procedures were used. First, concentrations of the elements observed in the top 6 inches of soil for each area sampled were tabulated. It soon became apparent that lead (Pb) is an excellent tracer for identifying former orchard land. From Pb analyses of 2 row crop areas (locations 6 and 10), 2 pastures (locations 6 and 8), and 1 woodland (location 8) it was suspected that these formerly supported orchards. This was confirmed by conversations with land owners. The values derived from these locations were omitted in developing average concentrations for the top 6 inches of soil in each area. The results, including averages and ranges, are given in table 9.

The second method used for establishing probable background concentrations was to average the concentrations of each element found in the top and bottom 6 inches of the 12-inch soil samples obtained from pastures and woodlands (excluding locations 6 and 8 for pastures and location 8 for woodlands). The results are shown in table 10. The values shown are not unlike the average values in table 9.

The third approach was based on the assumption that an array of concentrations representative of natural conditions would be distributed on arithmetic probability plots as one statistical population. Similarly, any man-induced values would produce another statistical population. Normal probability plots for As, Pb, Cu, Zn, and Fe are shown in figures 25 through 29 for the top six inches of soil for all uses.

The data for arsenic and lead (figures 25 and 26) clearly consist of two different populations in each case. The lower line of best fit represents naturally occurring concentrations of arsenic and lead. The range of values for these data is similar to those given in table 10. This suggests that the use of concentrations for pastures and woodland for background assessments is reasonable. The plots for Cu, Zn, and Fe in figures 27, 28, and 29 indicate a single population for each element. The implication is that on the average, practices of orchard horticulture have not influenced the concentration of these trace elements in the soils.

The source of As and Pb contributing to the increase of these elements above background levels in orchard soil is most likely lead arsenate. The correlation between them is shown in figure 30. A similar correlation for the background concentrations of Cu and Zn is shown in figure 31. This particular correlation, with a coefficient of 0.85, indicates that these elements have a natural association in undisturbed soils.

In the Cedar Lake watershed, the average background concentration for arsenic is about 8 ppm, and that for orchard soil is 29 ppm. From values reported for As by other investigators (Bowen, 1966; Wiersma et al., 1972; Jones and Hatch, 1945), it appears that arsenic concentrations vary from about 1 to 13 ppm in most soils. Values reported for orchard soils in Oregon (Jones and Hatch, 1937, 1945) were generally higher, varying in mean values from 20 to 56 ppm arsenic. These concentrations are not unlike those detected during this study.

The content of Pb in the non-fruit producing soils in the Cedar Lake watershed averages 18 ppm. The orchard soils averaged 91 ppm Pb with a range in the top 6 inches of soil of 34 to 230 ppm. Hinesly (1976) found 30 ppm Pb in silt loam used as an agriculture test plot in Champaign County, Illinois. Garcia et al. (1'974) found a range of 27 to 30 ppm Pb in Illinois strip mine soil. Soils in the vicinity of orchards in Oregon (Jones and Hatch, 1945) ranged from 3 to 20 ppm Pb but mean values for orchard land ranged from 118 to 376 ppm Pb. Bowen (1966) reports that the average Pb content of U.S. soils is 10 ppm. There is no doubt that the

Table 9. Average and Range of Concentrations of Heavy Metals in Soils

(Concentrations in ppm)

	As	Pb	Cu	Zn	Fe	Hg
Orchard soils	29	91	18	53	13,870	0.04
	13-17	34-230	10-32	41-87	7,000-22,000	0.03-0.06
Row crops*	9	22	13	43	13,825	0.03
	5-16	13-28	8-18	28-59	9,500-18,000	0.01-0.04
Pasture*	9	19	14	44	14,112	0.03
	7-12	12-23	9-18	31-62	10,500-21,500	0.02-0.05
Woodland*	9	17	13	40	12,470	0.03
	5-18	11-24	9-19	27-61	9,000-17,500	0.02-0.06

**Excluding former orchard soils*

high Pb values observed in the orchards during this study are man-induced.

The average concentration for Cu considered representative of natural conditions near Cedar Lake is 13 ppm, and that for orchard soils is 18 ppm. For Illinois conditions, Hinesly (1976) reports a range of 11 to 15 ppm Cu for'silt loam previously described. Strip mine soil ranged from 33 to 37 ppm Cu (Garcia et al., 1974).

The average Zn content considered background is 42 ppm. The average for orchard soil is 5 3 ppm. Hinesly (1976) reported the Zn content in an Illinois soil ranged from 66 to 71 ppm. For soils obtained from strip mines (Garcia et al., 1974) the range was 90 to 125 ppm Zn. Bear (1957) reported a Zn concentration range of 20 to 200 ppm for native soils in France.

The average content of Hg in terms of a background concentration is 0.03 ppm, and that for orchard soil is 0.04 ppm. If there is a difference between the Hg content in the soils, it is not perceptible. Wiersma and Tai (1974) reported a range of Hg from 0.05 to 0.10 ppm for soils in the eastern U.S. Gross (1976) found Hg in the sandstone and shale formations forming Cedar Lake ranging from 0.01 to 0.04 and 0.02 to 0.13 ppm, respectively. He concluded that the natural mercury content of these geologic formations correlated well with grain size, the fine-grained shales containing the most mercury and the coarse-grained sandstone containing the least mercury. D'Intri (1971) observed natural soils on a lake watershed with Hg contents ranging from 0.06 to 0.16 ppm, and Ruch et al. (1971) reported the Hg content of Illinois coal as averaging 0.18 ppm. The relatively low values of Hg in orchard soils seeem to rule out the spraying practices of the orchardists as major sources of mercury in the watershed.

Although the *average* values and ranges for each of the soils examined have been set forth in tabular form and compared with the findings of others, it is worthwhile to examine the inorganic pesticide residues in the soils having different uses at each of the 10 locations. In figures 32 through 35, the values for locations 2 and 3 were used interchangeably for row crop and woodland soil because of the proximity of the two locations.

As shown in figure 32, the row crop soil in location 10, the pasture soils in locations 6 and 8, and the woodland soils at location 8 exceed anticipated As concentrations. The average As content for background purposes is 8 ppm. More clear cut is the variation at these locations for Pb content as shown in figure 33. Here the row crop soils in locations 6 and 10 are substantially above the background average for Pb of 18 ppm. This is also the situation for pasture soils in locations 6 and 8 and for a woodland soil at location 8. All the soils mentioned here formerly supported orchards. Where samples were not obtained for row crop soils and, in one instance for woodland soil, these types were not in the vicinity of the orchard selected for that location.

Although the general inference from this discussion has been that orchard practices have not influenced the content of copper and zinc in soils, the values presented in figures 34 and 35 indicate that differences do occur in some areas. For Cu and Zn content, as shown in figures 34 and 35, respectively, there are some soils with higher concentrations than the average of 13 ppm Cu and 42 ppm Zn reported

Table 10. Probable Background Concentrations of Heavy Metals

(Concentrations in ppm)

	Average	Range	Number of samples
Arsenic	8	2-15	30
Lead	18	9-29	30
Copper	13	6-27	30
Zinc	42	26-62	30
Mercury	0.03	0.01-0.06	28
Iron	14,280	9,000-20,000	30

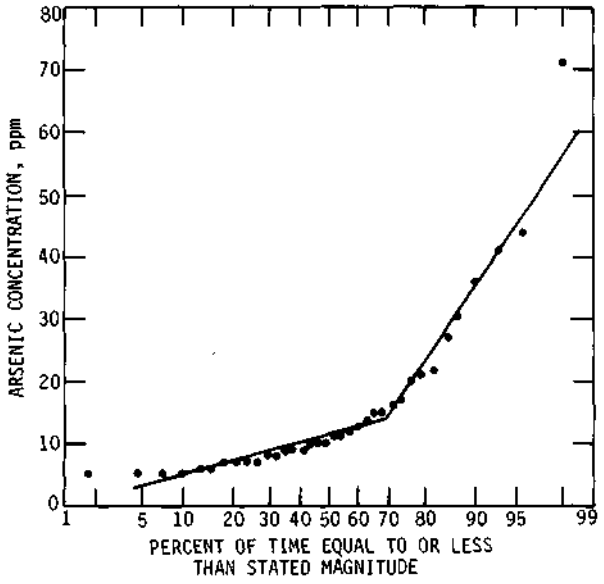

Figure 25. Probability of data distribution for arsenic in soils

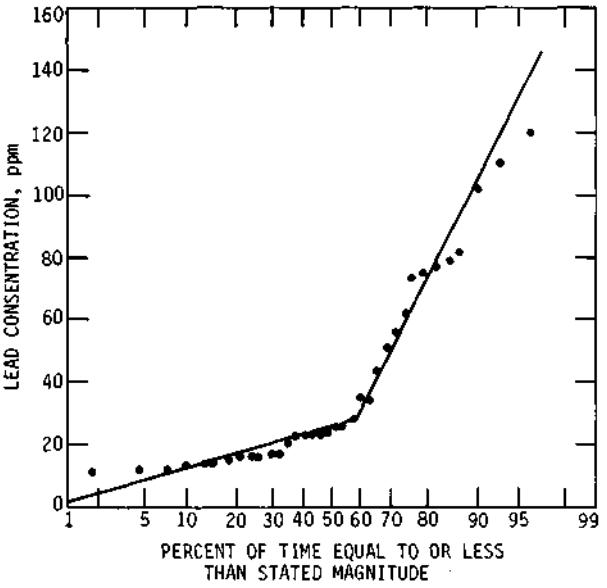

Figure 26. Probability of data distribution for lead in soils

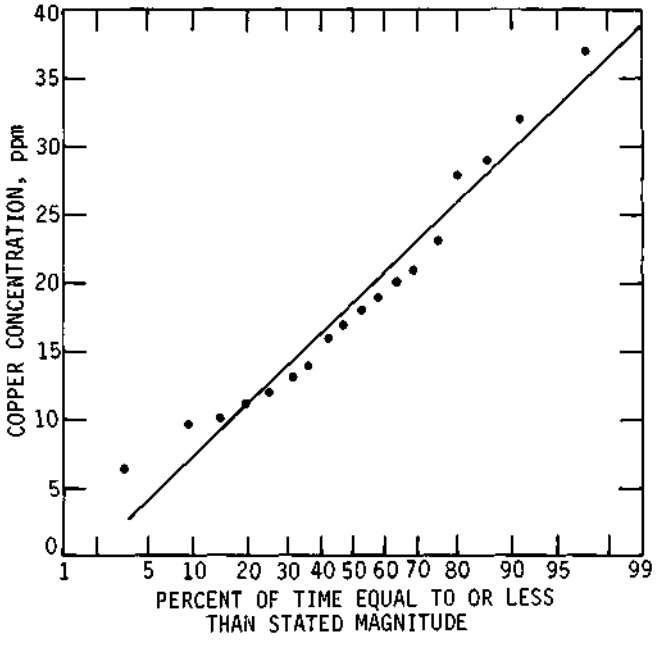

Figure 27. Probability of data distribution for copper in soils

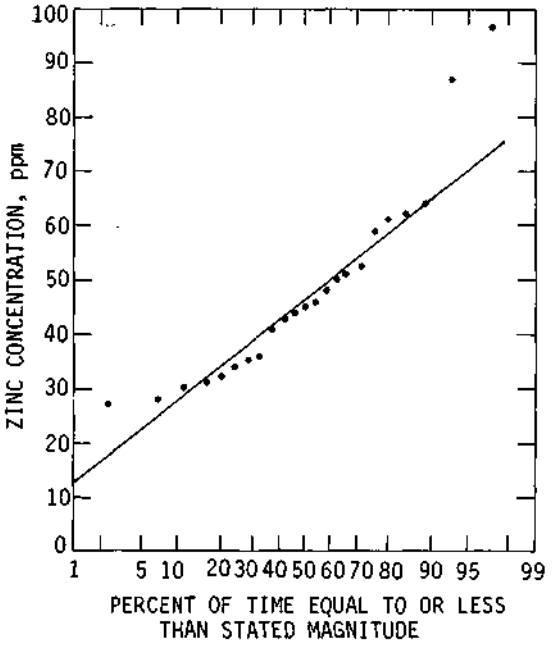

Figure 28. Probability of data distribution for zinc in soils

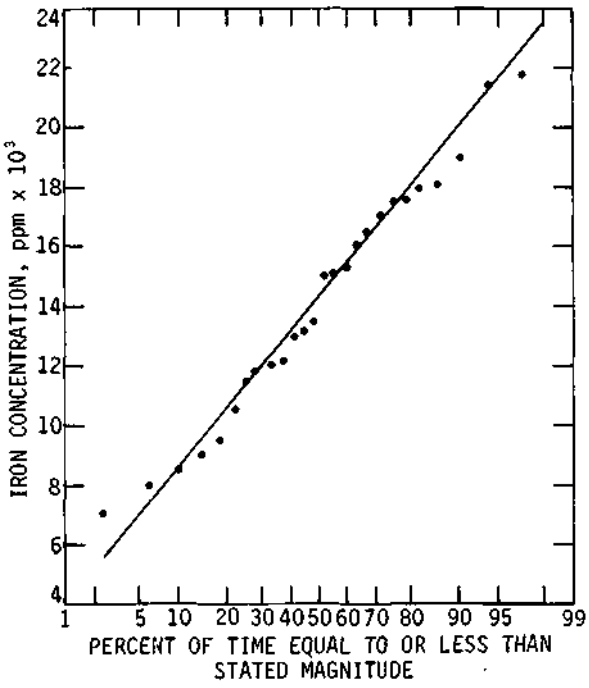

Figure 29. Probability of data distribution for iron in soils

here. Orchard soils in locations 5, 6, and 9 and the former orchards now serving as cropland, pastures, and woodland in locations 6 and 8 display Cu contents significantly higher than the average background concentration. This is not the case for the majority of the 10 orchards. A similar situation exists for Zn concentrations. Of particular interest are the orchard soils in location 5 and the former orchard soil now serving row crops in location 6.

The data gathered to define the content of inorganic pesticide residues in soils within the Cedar Lake watershed have been evaluated and discussed. However, the magnitude of the residues is likely to vary spatially, as well as within the soil profile, for orchard soils. Some limited sampling and analyses are reported here to demonstrate this.

On three occasions soil samples were taken in the vicinity of fruit trees at 1) the dripline, 2) midway from the dripline to the tree trunk, and 3) near the tree trunk. The trace elements of these samples are given in table 11. Higher concentrations were expected at the dripline. This was the case for Pb at location 9 but not at the other locations. At each location, the concentrations of Pb, Cu, and Zn were generally less near the tree trunks than at the midway sites or the driplines.

On two occasions, soil samples were taken in orchards to determine the vertical distribution of trace elements at 2-inch intervals in a 6-inch deep core. They were obtained from locations 8 and 9. The results are shown in table 12.

The concentrations of Pb decreased with increasing soil depth. This appears to be the case also for Cu and Zn concentrations but to a lesser extent. Composited samples from locations 8 and 9 at these sampling sites would average 142 and 91 ppm, respectively, for Pb content. These are about equivalent to the concentrations of the 2- to 4-inch sections of the cores.

Heavy Metals in Streams

The samples collected to reflect the concentrations of Pb, Cu, Zn, Hg, and Fe being conveyed by streams were quite limited. Because of drought conditions, streamflows were extremely low and the suspended sediment being transported was minimal. On one occasion, samples were collected from the watershed streams at a time of high flow. During the study the water level of Cedar Lake was lowered sufficiently to expose deposits of sediment at the mouth of three streams, and this provided an opportunity to obtain samples of these deposits on one occasion.

The suspended sediment samples were analyzed in a conventional manner producing concentration values in terms of milligrams per liter (mg/1). This expression is not compatible with the values (ppm) developed for soils and sediment. Conversions of the results from mg/1 to ppm by calculation are shown in table 13. The relatively high values are consistent with observations made on the Spoon River in Illinois. A single sample from each stream, as is the case here, does not provide an ample basis for discussing these results. However, it is worthwhile to compare them with the results for the stream deposits obtained at the mouth of three streams, as shown in table 14. The alluvial deposits represent those put in place since the formation of Cedar Lake and should be a representative composite for the sediments conveyed by the streams since that time. The concentrations of Pb, Cu, Zn, Fe, and Hg shown are similar to the average natural concentrations noted previously for soils in the watershed, i.e., 18, 13, 42, 14,280, and 0.03 ppm, respectively. The implication is that the impact of orchard soils with higher trace element densities is not detectable in the stream sediment deposits.

Heavy Metals in Lake Sediments

Samples were obtained from the bottom sediments of Cedar Lake, Little Cedar Lake, and the Alto Pass Reservoir. Samples were also collected for examination of the interstitial waters of the bottom sediments of Cedar Lake as well as the mud-water interface of that lake. Each sample was analyzed for lead, copper, zinc, mercury, and iron content.

The bottom mud samples were collected in triplicate from four locations in Cedar Lake (see figure 3) and two locations each on Little Cedar Lake and the Alto Pass Reservoir, one near the dams and one in the upper reaches of each impoundment.

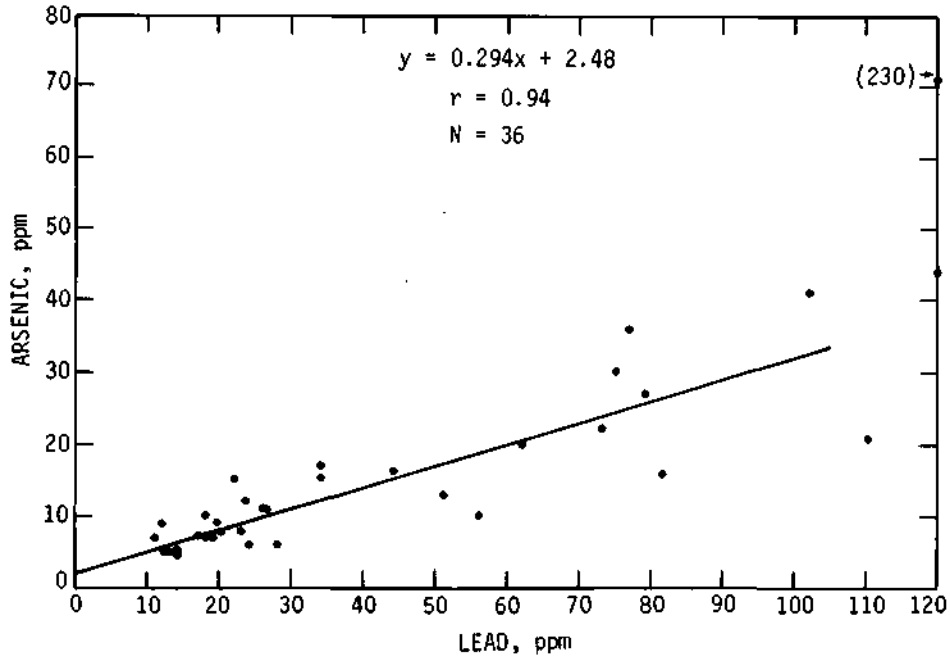

Figure 30. Relationship of arsenic and lead in soils

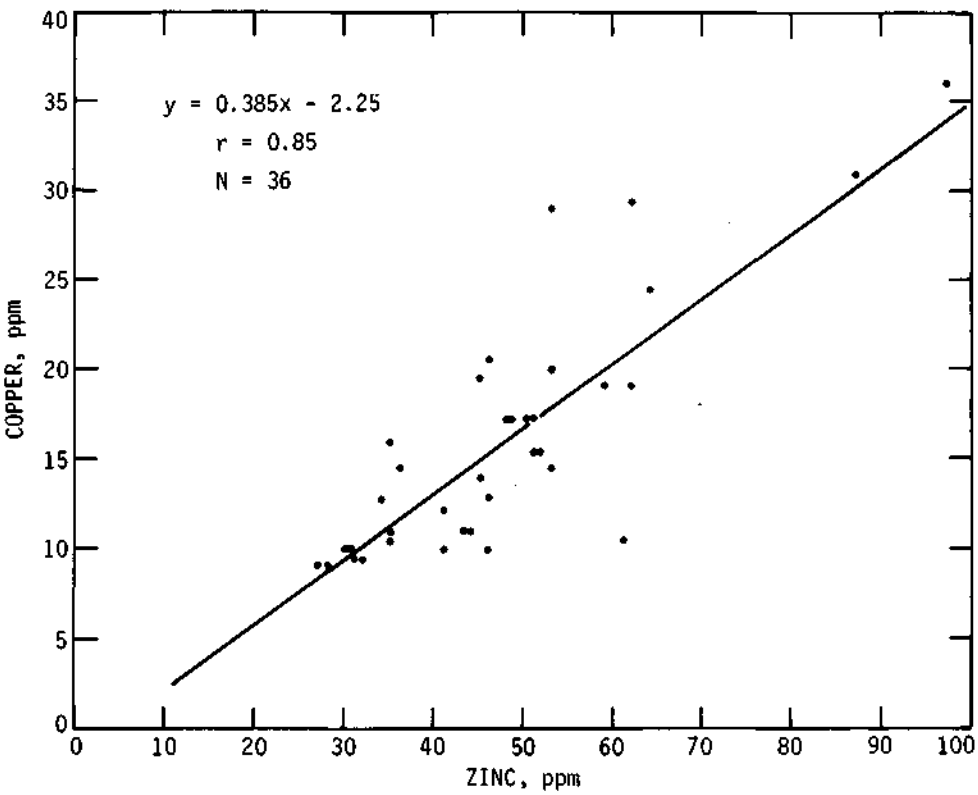

Figure 31. Relationship of copper and zinc in soils

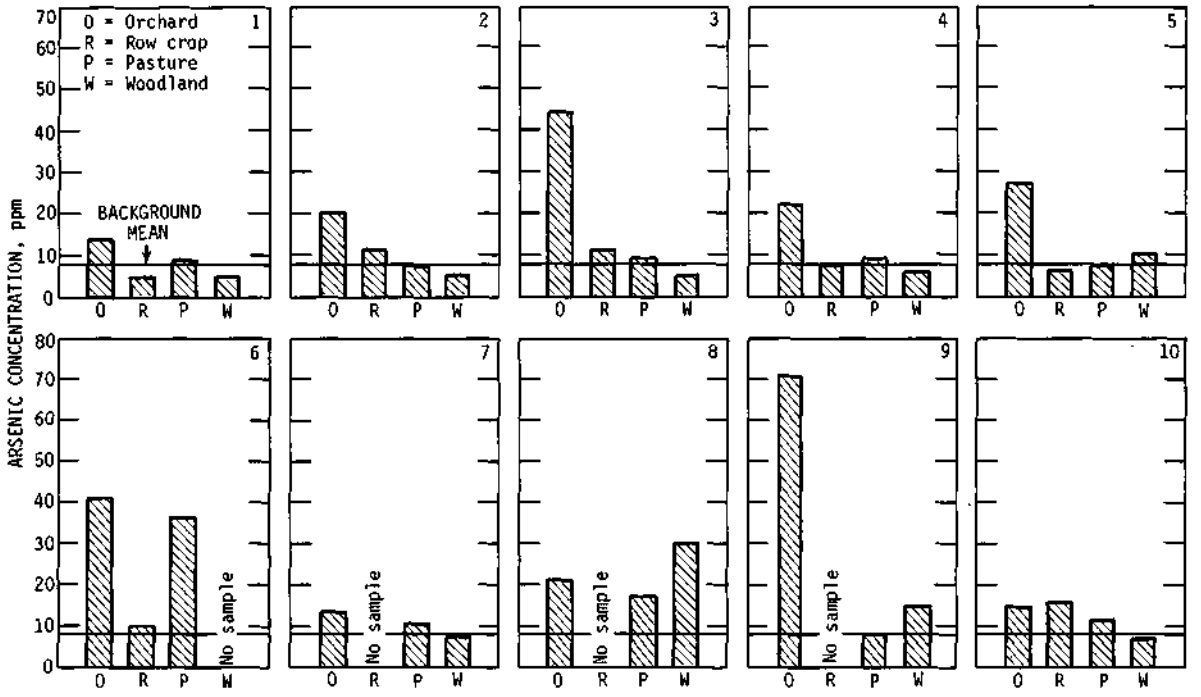

Figure 32. Arsenic concentrations in top 6 inches of soil used for different purposes

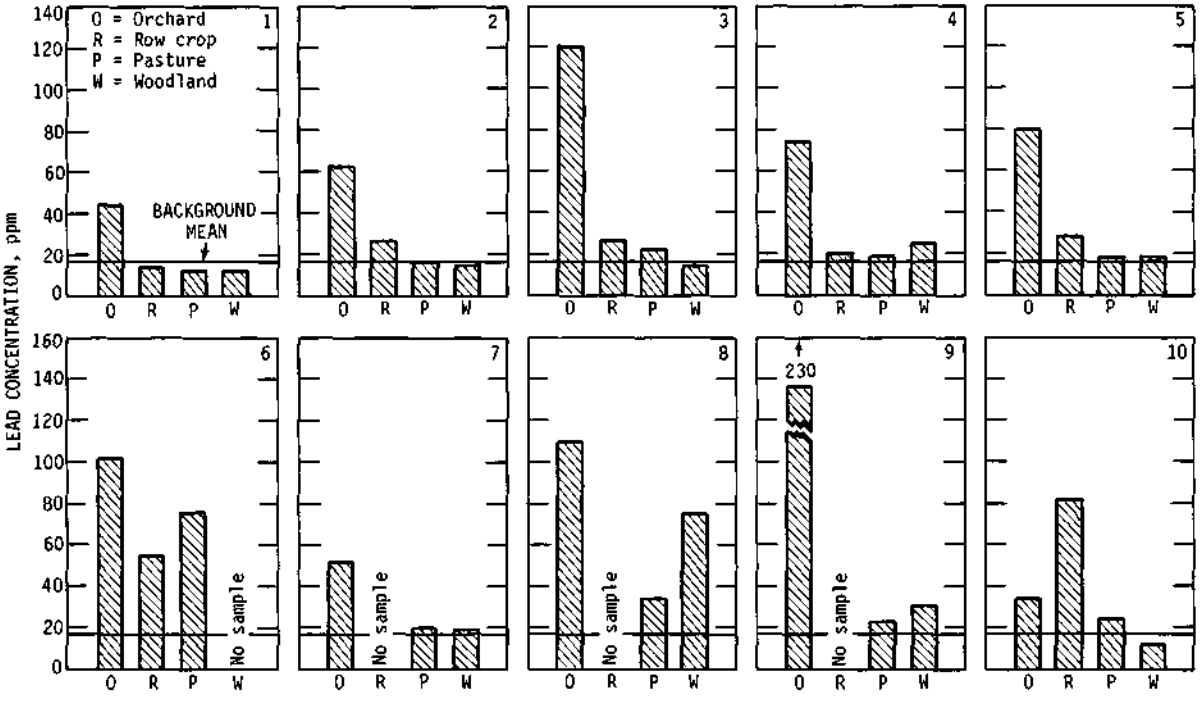

Figure 33. Lead concentrations in top 6 inches of soil used for different purposes

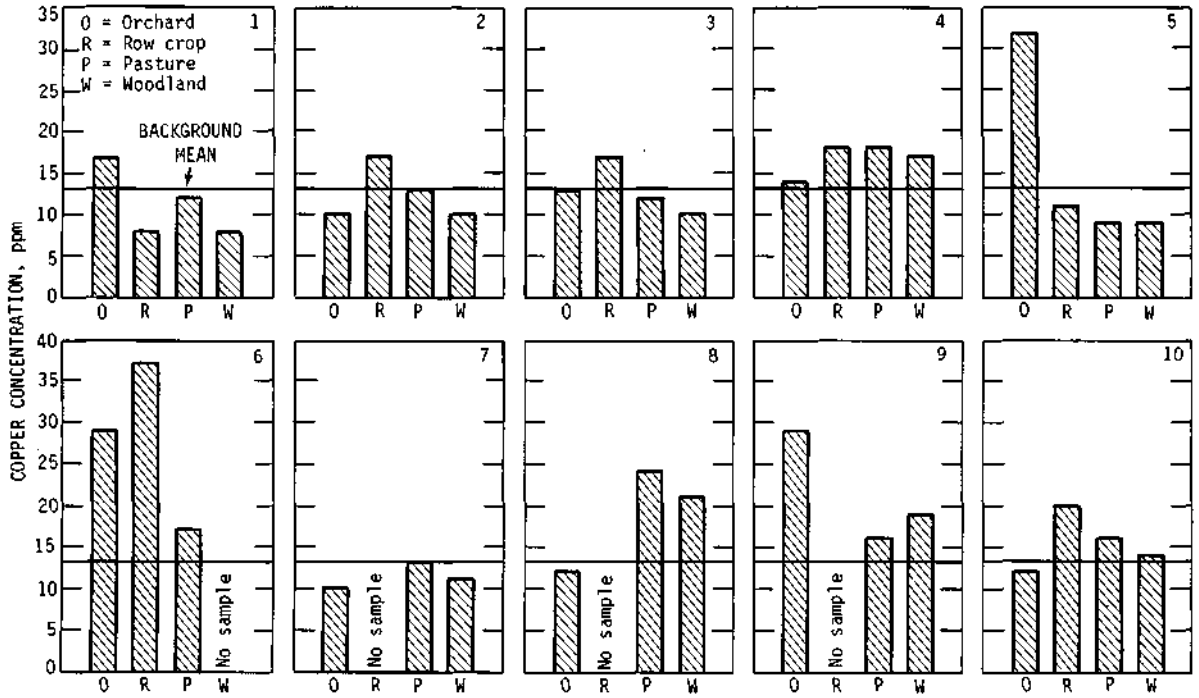

Figure 34. Copper concentrations in top 6 inches of soil used for different purposes

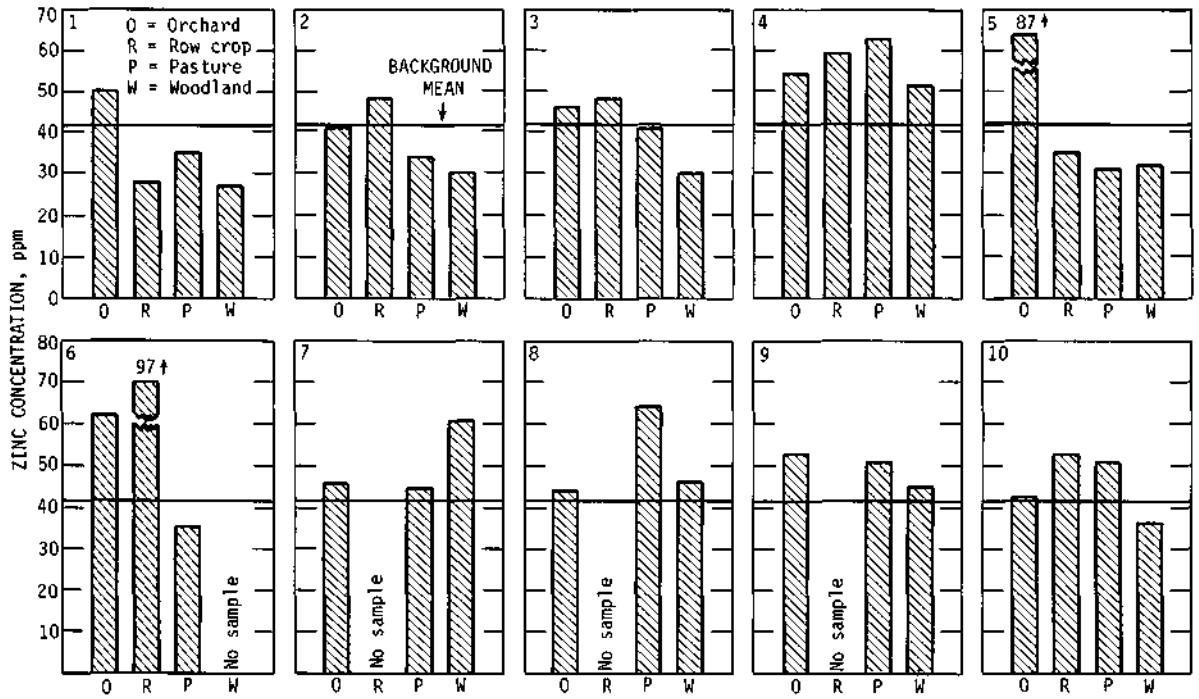

Figure 35. Zinc concentrations in top 6 inches of soil use used for different purposes

The sediments of Little Cedar Lake and Alto Pass Reservoir, representative of a 7-year old and a 9-year old impoundment, respectively, were chosen for comparison with those of 2-year old Cedar Lake. The average concentrations in these sediments are summarized in table 15.

In every case except for Hg and Fe, the trace element concentrations in the sediments of the older impoundments exceeded those in Cedar Lake. Also, except for station 4 on the Cedar Creek leg of the lake, the concentration of trace elements (except Hg) in the sediments of that lake exceeded those of the natural soils of the watershed. The general trend of the concentrations, as shown in table 15, is an increasing content of trace elements in sediments with increasing age of the impoundments.

The average content for Pb in Cedar Lake sediments ranged from 15 to 26 ppm. These values are comparable to those reported by Butts et al. (1976) for Dolan Lake in Hamilton County, Illinois, a range of 17 to 29 ppm. The streams in the vicinity of Dolan Lake had sediment contents ranging from 10 to 30 ppm Pb. The sediments of Peoria Lake, reported by Collison and Shimp (1972), had 146 ppm Pb. Lead content in the sediments of the Fox Chain of Lakes in Illinois ranged from 34 to 71 ppm (Kothandaraman et al., 1977). In comparison with the mean values for lead for other bodies of water, the sediments of Cedar Lake are not severely degraded. However, the difference between the concentration observed *in situ* and the background concentrations of the soils in the watershed suggest that Pb is accumulating in the sediments.

The range of average Cu concentrations in the Cedar Lake sediments is 10 to 19 ppm. These values are not inconsistent with those cited by other investigators (Schacht and Matsunaga, 1975; Mathis and Cummings, 1973; Kothandaraman et al., 1977). The older impoundments in the vicinity of Cedar Lake show an average Cu concentration range of 16 to 34 ppm. The same pattern appears for Zn content. In this case the range for Cedar Lake is 35 to 54 ppm Zn; for the older impoundments, on the average, it is 42 to 95 ppm Zn. Again, these values are comparable with those found by other investigators (Proctor et al., 1975; Colbert et al., 1975; Kothandaraman et al., 1977).

Concentrations of mercury in lake sediments in the Cedar Lake watershed ranged from 0.01 to 0.05 ppm, and these were exceeded tenfold in other parts of the country (Baumgartner et al., 1977; Sustar and Wakeman, 1977; Rickert et al., 1977).

With the exception of mercury, all heavy metals appear to be accumulating in the sediments of Cedar Lake. This inference is based principally on the differences in concentrations of trace elements between the natural soils of the watershed and the sediments of the lake. In two core samples from the sediments of Little Cedar Lake a segment of the original bottom was obtained and analyzed. The concentrations observed were similar to those of the natural soils as shown in table 16.

Table 11. Heavy Metals in Orchard Soils Beneath a Fruit Tree

(Concentrations in ppm)

Location	Pb	Cu	Zn	Fe	Hg
8 D*	132	14	76	12,000	0.04
M	160	20	79	12,000	0.04
T	61	13	70	16,000	0.04
9 D	510	50	68	13,000	0.06
M	442	47	74	15,000	0.07
T	300	37	69	15,000	0.06
10 D	70	10	52	10,000	0.06
M	71	8	58	13,000	0.10
T	34	8	50	16,000	0.10

**Dripline = D; Midway = M; Trunk = T*

Table 12. Heavy Metals in Profile of Orchard Soils

(Concentrations in ppm)

Location	Depth (inches)	Pb	Cu	Zn	Fe	Hg
8	0–2	200	16	72	9,100	0.01
	2–4	130	14	61	10,200	0.02
	4–6	96	10	39	9,300	0.04
9	0–2	176	29	62	12,000	0.04
	2–4	86	13	47	10,600	0.04
	4–6	17	15	46	15,800	0.04

It is not unreasonable to expect from the comparisons observed for Little Cedar Lake and the values given in table 15 for the Alto Pass Reservoir, that these are useful examples for estimating the rate of accumulations now being experienced by Cedar Lake sediments.

The principal reason for examining the interstitial waters of the bottom muds and the mud-water interface is to determine the extent that heavy metals are being solubilized in the bottom sediments. It is assumed that the trace elements will concentrate in these liquors under anaerobic conditions and then, during periods of lake overturn, will be distributed throughout the lake waters. The concentrations observed for interstitial water are shown in table 17. The results are unexpected in that higher concentrations of Pb were detected in Cedar Lake than in Little Cedar Lake in spite of the fact that the sediments of Little Cedar Lake were higher in Pb than in Cedar Lake. In retrospect more than one sample per lake station should have been collected.

Samples at the mud-water interface were collected on four occasions from the four Cedar Lake stations and the two Little Cedar Lake stations. The sampling periods were scheduled for spring (May 12), midsummer (July 29 and August 24), and late fall (October 28). These periods were selected to coincide with aerobic and anaerobic conditions occurring at the bottoms of the lakes (Roseboom et al., 1978).

Table 13. Computed Heavy Metals on Suspended Solids in Streams

(Concentrations in ppm)

	Pb	*Cu*	*Zn*	*Fe*	*Suspended solids (mg/l)*
West Cedar Creek*	139	91	117	17,340	184
West Cedar Creek**	<80	144	176	30,000	129
East Cedar Creek	80	124	88	15,360	253
Clay Lick Creek	118	133	155	28,500	135
Mill Creek	<154	231	169	27,800	65
Poplar Camp Creek	<80	144	128	34,800	125

**Upstream of Alto Pass Reservoir*
***Downstream of Alto Pass Reservoir*

Table 14. Heavy Metals in Alluvial Sediments of Streams

(Concentrations in ppm)

	Pb	*Cu*	*Zn*	*Fe*	*Hg*
Clay Creek	22.0	11.0	41.0	11,000	0.02
Mill Creek	17.0	10.0	35.0	10,500	0.01
Poplar Camp Creek	17.00	10.0	35.0	15,000	0.02

Table 15. Average Concentrations of Heavy Metals in Lake Sediments

(Concentrations in ppm)

Location	*Pb*	*Cu*	*Zn*	*Hg*	*Fe*
1 Cedar Lake	21	14	53	0.01	18,700
2 Cedar Lake	24	19	54	0.03	20,800
3 Cedar Lake	26	18	54	0.03	17,800
4 Cedar Lake	15	10	35	0.03	9,300
5 Little Cedar Lake	45	29	95	0.03	26,500
6 Little Cedar Lake	45	31	89	0.05	26,500
7 Alto Pass Reservoir	61	34	77	0.03	23,000
8 Alto Pass Reservoir	35	16	42	0.02	13,500

Table 16. Comparison of Original and Present Sediments in Little Cedar Lake

(Concentrations in ppm)

	Pb	*Cu*	*Zn*	*Hg*	*Fe*
Natural soil	18	13	42	0.03	14,200
Original bottom, Little Cedar Lake	22–25	12–14	32–42	0.02–0.03	10,500–14,300
Sediments, Little Cedar Lake	40–54	21–46	64–117	0.02–0.06	18,000–36,000

Table 17. Concentrations of Heavy Metals in Interstitial Waters

(Concentrations in micrograms per liter)

	Location	Pb	Cu	Zn	Fe	Hg
1	Cedar Lake	52	11	76	300	0.43
2	Cedar Lake	62	<10	74	2,340	0.14
3	Cedar Lake	52	<10	55	12,200	0.16
4	Cedar Lake	<10	17	61	800	0.75
5	Little Cedar Lake	<10	<10	44	13,000	0.32
6	Little Cedar Lake	<10	<10	46	17,500	0.21

An earlier study by Hullinger (1975) of the waters of the Alto Pass Reservoir was used for comparison purposes. It was assumed that the trace metal content of the waters of that 9-year-old impoundment would not be unlike Cedar and Little Cedar lake waters. Table 18 shows the average concentrations at the mud-water interface of the two lakes on July 29 when the waters of Cedar Lake were devoid of dissolved oxygen from the 16-foot depth to the lake bottom (see figure 12). The Pb, Cu, and Zn concentrations at the mud-water interface were generally two to three times higher than those of the overlying waters (Alto Pass Reservoir data). This demonstrates that trace elements in the sediment do become solubilized and can be distributed within the lake waters. Wang (1975) reported that significant changes in ammonia nitrogen, silica, alkalinity, and nitrates also occur at the mud-water interface of an impoundment in the absence of dissolved oxygen. These facts support the view that a lake's bottom muds can alter the quality of its waters.

Figure 36 shows the seasonal effects on the average concentrations of trace elements at the mud-water interface of Cedar and Little Cedar Lakes. It is apparent that July was a critical month, as with the exception of iron, all concentrations were maximized on that date. After the lakes destratified in the fall, the concentrations on October 28 were comparable with those anticipated through the lake waters, concentrations not unlike those in table 18 for the Alto Pass Reservoir.

Summary

In Illinois the acreage in orchard cultivation is not great, but the use of pesticides is intensive. The inorganic pesticides that are being used, or that have been used, by orchardists leave residues consisting mainly of arsenic, lead, copper, zinc, and mercury. These are virtually nondegradable and their translocation is most often initiated by soil movement. In the orchards observed, continuous sod cover was commonly practiced, so that excessive soil erosion is unlikely.

Background concentrations for the soils examined (orchards, cropland, pasture, and woodland) for arsenic, lead, copper, zinc, mercury, and iron were, respectively, 8, 18, 13, 42, 0.03, and 14,280 ppm. In all orchard soils examined the concentrations of arsenic and lead exceeded background concentrations. The average for arsenic was 29 ppm; that for lead was 91 ppm.

Lead is an excellent tracer for identifying orchard soil in the study area. Former orchard land now supporting row crops and pasture were initially identified as former orchards by their lead content. The row crop areas on former orchard lands are a significant source of soil movement.

The significance of higher than normal concentrations of copper and zinc in some soils (orchards, row crop, and pasture) is not clear. Copper chemicals are not being used extensively in the area and records on past use are meager.

Mercury concentrations in orchard soils did not significantly differ from background concentrations.

The limited sampling of suspended sediment in streams was not useful in determining the extent of inorganic residue movement via waterways. Recent deposits (2-year old) at the mouth of major tributaries suggest the suspended sediment being conveyed originated from soils with background concentrations. However, in examining the bottom sediments it is apparent that an accumulation of inorganic residues, with the exception of mercury, is occurring within the lake system. The sediments of Cedar Lake in terms of trace element content are not significantly degraded, but there is evidence, based on observations in Little Cedar Lake and the Alto Pass Reservoir, that trace element concentrations are likely to increase with increasing age of the impoundment.

Table 18. Comparison of Overlying Water with Mud-Water Interface

(Concentrations in micrograms per liter)

Location	Pb	Cu	Zn	Hg
Alto Pass Reservoir	21	7	55	0.40
Cedar Lake*	38	15	90	0.25
Little Cedar Lake*	55	32	177	0.27

**Mud-water interface samples*

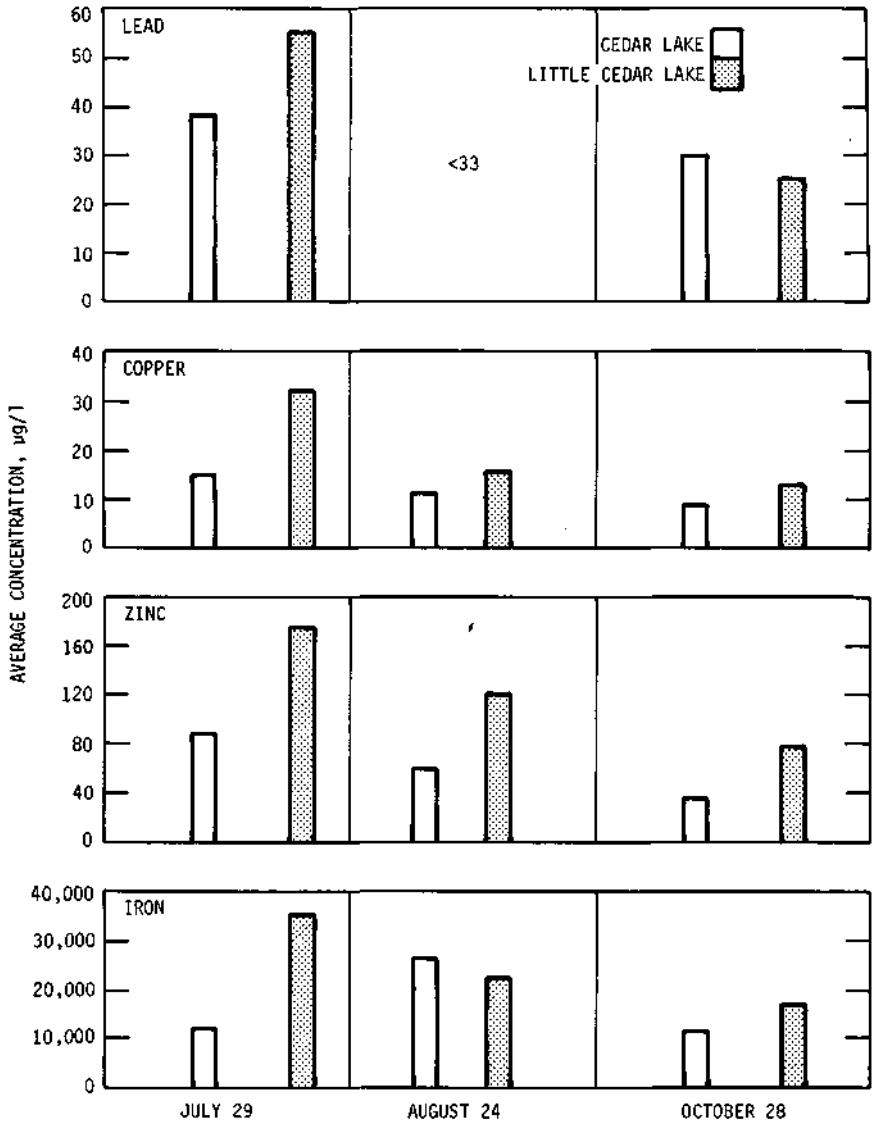

Figure 36. Heavy metals concentrations at mud-water interface

Examinations at the mud-water interface show that trace elements accumulating in the bottom sediments do become solubilized and can be transferred to the lake waters.

Organic Pesticides

The types of organic pesticides that have been used by orchardists include organochlorines, organophosphates, and carbamates. The extreme stability or persistence of the organochlorine pesticides has led to restrictions on their use. They have been replaced by the more unstable organophosphates and carbamates. The organochlorine pesticides that had been commonly used before their ban include DDT, dieldrin, endrin, and lindane.

DDT is degradated to its metabolites DDE and DDD. DDE is a product of aerobic activity; DDD is generally produced under anaerobic conditions. Lichenstein et al. (1960) observed that DDT will degrade in soil on an average of 50 percent (range of 26 to 78 percent) of its initial concentration during a 3-year period. Dieldrin, on the other hand, is probably the most persistent of the organochlorines (Terriere and Ingalsbe, 1953).

The degradation of pesticides may be 'complete' with inorganic by-products, or 'partial' as in the case of DDT with intermediate organic by-products. The metabolites of DDT are less toxic to man, domestic animals, and wildlife than the original product (Spencer, 1967). However, the degradation products of carbamates may be many times more toxic

than the parent compound (Teledyne Brown Engineering, 1972). Available information on degradation rates, and particularly on the mechanisms involved and the by-products formed, for organophosphates and carbamates is limited to laboratory studies. How confidently these findings can be extrapolated to natural conditions is not known. Without more precise knowledge there should not be complete comfort from the fact that these compounds are not often detected in the environment. However, we can say that chlorinated hydrocarbons persist in the environment in terms of years, whereas organophosphates and carbamates disappear within a matter of a few weeks or months.

The aquatic hazard from pesticides depends largely on the persistence of the pesticides. Although the most persistent types are no longer being used, their mere persistence necessitates periodic evaluation of their presence in the aquatic environment.

In the absence of wind drift during application, the major pathway for orchard pesticides to enter streams and lakes is by runoff and erosion. During the study of orchards in Oregon (Tierrere et al., 1966) and in New York (Kuhr et al., 1974), it was found that DDT is bound to the soil and very little is lost to wind drift, water runoff, or downward leaching. In water runoff from land there are two phases, the water phase and the sediment phase. Pionke and Chesters (1973) found that the ratio of the concentration of organochlorine pesticide in the runoff water to that absorbed by suspended soil particles was as great as 1 to 1000 for DDT and dieldrin. This emphasizes the importance of soil movement in pesticide translocations.

When a pesticide is introduced from land, either absorbed on sediment or in solution, to the aquatic environment it is subjected to a completely different set of environmental conditions. Sediment-water interactions are, perhaps, the key to the distribution of pesticides within the waters of lakes. Where organic material and the amorphorus mineral content of the sediments are high, a pesticide may become more strongly absorbed to the sediment and relatively resistant to microbial degradation. On the other hand, under anaerobic conditions the degradation rates, stimulated by a larger more efficient microbial population, are enhanced.

Pesticides introduced from sediments into the water column may be concentrated by adsorption and biomagnifications. Sorption by algae and aquatic vegetation is rapid, achieving concentrations of 100 to 10,000 times the DDT and parathion concentrations in water. The extent of lindane adsorption has been correlated directly with increase in algae concentration (King et al., 1969).

During this study, analyses were performed for 11 organic pesticides in the manner previously described. The pesticides for which determinations were made are listed in table 19. Only DDT, DDE, DDD, dieldrin, and endrin were detected in the samples within the detection limits shown in table 3. Concentrations are reported on a dry weight basis and expressed as parts per billion (ppb) for soils and sediment and micrograms per liter (jug/l) for water. The DDT reported is p,p DDT.

Organic Pesticides in Soils

DDT was detected in all soil samples gathered from orchards except one. Dieldrin was detected in three orchard samples and two row crop samples but not in any soils supporting pastures or woodlands. Endrin was detected in one orchard soil (location 7). It is likely that dieldrin is more widely distributed in the Cedar Lake watershed than reported here. Samples from selected orchard soils and the three stream alluvial deposits were extracted by the use of a florisil separation column. The technique revealed concentrations of dieldrin at most of the sites reported here, but at concentrations generally less than 200 ppb in soils and 40 ppb in sediment.

As in the case of heavy metals concentrations, it was desirable to establish a background concentration for organic

Table 19. Organic Pesticides Analyzed

Compound	*Use*	*Type*	*Detected* *Yes*	*No*
Lindane	Insecticide	Organochlorine		X
DDT, DDE, DDD	Insecticide	Organochlorine	X	
Dieldrin	Insecticide	Organochlorine	X	
Endrin	Insecticide	Organochlorine	X	
Kelthane	Miticide	Organochlorine		X
Malathion	Insecticide	Organophosphate		X
Parathion	Insecticide	Organophosphate		X
Guthion	Insecticide	Organophosphate		X
Tedion	Miticide	Chlorinated sulfone		X
Captan	Fungicide			X
Folpet	Fungicide			X

Table 20. Pesticides Found in 199 Noncropland Soil Samples in 11 States in 1969*

	Number of positive samples	Percent of sites positive	Concentration (ppb) Mean	Range
DDT	18	9	10	<10-230
DDE	27	14	10	110-310
Dieldrin	8	4	<10	<10-90

*From Wiersma et al., 1972

Table 21. DDT in Cedar Lake Watershed Soils

	Number of samples analyzed	Number of positive samples	Percent of samples positive	Concentration (ppb) Mean	Range
Orchard	20	19	95	1098	10-2730
Row Crop*	12	2	17	14	7-20
Pasture*	16	3	13	11	5-20
Woodland*	14	4	28	28	5-60

*Excluding former orchard soils

pesticides in soils. For comparisons, reliance was placed on the work by Wiersma et al. (1972), in which samples of soil from noncroplands in 11 states were examined. Organic pesticide information pertinent to this study is given in table 20. The mean concentration for DDT and its metabolite DDE is 10 ppb with ranges of <10 to 310 ppb. For the Cedar Lake watershed, as shown in table 21, the mean concentration for DDT in nonorchard soils ranges from 11 to 28 ppb with an overall concentration range of 5 to 60 ppb. Also, 95 percent of the orchard sampling sites were positive for DDT, whereas 13 to 28 percent of the nonorchard soils contained DDT concentrations. On the basis of these data it is not unreasonable to assume a mean range of 10 to 20 ppb DDT in nonorchard soils for background levels. Orchard soils averaged 1098 ppb DDT and 2475 total DDT (DDT plus DDE).

The concentrations of DDT found in the top 6 inches of soil for orchards, row crops, pastures, and woodlands at different locations are shown in table 22. The concentration range for orchard soil was 100 to 2730 ppb. Although soil type, compound structure, use, and organic content will often govern the concentrations of organochlorine pesticides in soil, it is likely that the variations shown in table 22 may reflect the age of the orchards. Concentrations of DDE were detected in all orchard soils but DDD was not. Nor was DDD detected in any other type of soil use. The DDT concentration in orchard soils is considerably higher than the assumed background concentration range of 10 to 20 ppb. However, in comparing the Cedar Lake orchard soils with those of southern Indiana before the ban on DDT (Murphy et al., 1964) the DDT content seems less significant. The DDT content in the soils of 35 orchards in southern Indiana ranged from 3298 to 67,219 ppb with an average of 24,162 ppb.

The observations of Lichenstein et al. (1960), previously mentioned, which show that DDT will degrade on an average of 50 percent during a 3-year period may explain some of the differences between the observations of DDT content in orchard soils in southern Indiana compared with those reported here for southern Illinois. It is possible that the DDT contents in southern Indiana may now be comparable with those in the Cedar Lake watershed.

One soil profile was examined for organic pesticide content for this study. The results are shown in table 23. As previously documented, the concentrations of organic pesticides diminish with increasing soil depth, ranging from 1860 ppb DDT in the upper 2 inches to 400 ppb in the 4- to 6-inch layer of soil.

The average concentrations of DDT in the soils of the orchards in this study were also compared with the average DDT content in soils during 1969 for other types of crops in the U.S., as shown in figure 37. Here the average concentration of DDT in Cedar Lake orchard soils exceeded, considerably, that of other soils supporting cotton, corn, vegetables, etc.

From the analytical procedures used in this study, dieldrin was detected in three orchard soils (locations 1,5, and 10) at concentrations in the top 6 inches of soil of 330,490, and 20 ppb, respectively. Dieldrin was also found in the top 6 inches of soil from row crop lands (locations 3 and 4) at concentrations of 30 and 24 ppb, respectively. It was not detected in any pasture or woodland soils.

The persistence of dieldrin in soils may pose problems other than those related to the aquatic environment. The

Table 22. Concentrations of DDT in Top 6 Inches of Soil

(Concentrations in ppb)

Location	Orchard	Cropland	Pasture	Woodland
1	380	ND	ND	ND
2	1850	No sample	8	5
3	2730	7	ND	No sample
4	250	ND	ND	9
5	730	20	ND	60
6	1950	700*	2560*	No sample
7	240	No sample	ND	ND
8	540	No sample	ND	260*
9	2210	No sample	ND	ND
10	100	1400*	20	ND

** Formerly orchards*
Note: ND = not detected

conversion of orchard land for other uses is a common practice. A study by Moubry et al. (1968), of former orchards being used for pastures in Wisconsin concluded that dieldrin residues in the milk produced from herds on the pastures was directly related to the history of dieldrin use in the orchards. This prompted the state of Wisconsin to prohibit dairy animals, or any animals being finished for slaughter, to graze in operating or abandoned orchard areas having a history of dieldrin application. The dieldrin concentrations on the former orchard pastures in Wisconsin ranged from 334 to 954 ppb; DDT ranged from 1890 to 1930 ppb. The soil sample consisted of composited 4-inch cores.

Endrin was found in the top 6 inches of one orchard at location 7. The concentration was 470 ppb. Endrin was not detected in any other soil sample regardless of use.

Organic Pesticides in Streams

Analyses were performed for organic pesticide concentrations in the waters and suspended sediment of Poplar Camp, Clay Lick, Mill, and Cedar Creeks. Organic pesticides were not detected. The persistence of organic pesticides in river water has been examined (Eichelberger and Lichtenberg, 1971) in the laboratory. The data demonstrate the relative persistence of organochlorines to that of organophosphates and carbamates. There was no diminishment in organochlorine concentrations over an 8-week period whereas, with the exception of ethion, all the other types of organic insecticides degraded within that time, and most within 2 weeks. However, in spite of their persistence, organochlorines were not detected in the waters of the major streams in the Cedar Lake watershed.

Samples of alluvial deposits at the mouth of three streams at their entrances to Cedar Lake revealed DDT, DDE, and DDD concentrations. The concentrations found (June 25, 1976) are summarized in table 24. These sediments have accrued since the dam closure for Cedar Lake in 1974.

The average DDT concentrations of 9 and 5 ppb for Clay Lick and Poplar Camp Creeks, respectively, are within the lower range of background levels. Those sediments deposited by Mill Creek were higher, averaging 23 ppb in DDT content. The higher values for Mill Creek become more evident when considering total DDT concentrations, i.e., the sum of the p,p DDT, DDE, and DDD. In this case, the average values were 62, 229, and 68 ppb, respectively, for Clay Lick, Mill, and Poplar Camp Creeks. The differences in concentrations for Mill Creek noted here were not evident for heavy metals (see table 14).

In about 165 sediment samples collected from five rivers in Ohio, Waldron (1974) found two samples containing DDT with concentrations of 5 and 16 ppb, four samples containing DDE residues ranging from 2 to 35 ppb, and 17 positive samples for dieldrin averaging 6 ppb, with concentration ranges from 1 to 16 ppb. He concluded that river sediments showing positive occurrences of organochlorine residues are negligible and insignificant. These conclusions are not applicable to the sediment deposits at the mouth of Mill Creek.

Table 23. Organic Pesticides in Profile of Orchard Soils, Location 8

(Concentrations in ppb)

Depth (inches)	DDT	DDE	DDD
0-2	1860	1980	ND
2-4	1440	1330	ND
4-6	400	430	ND

Note: ND = not detected

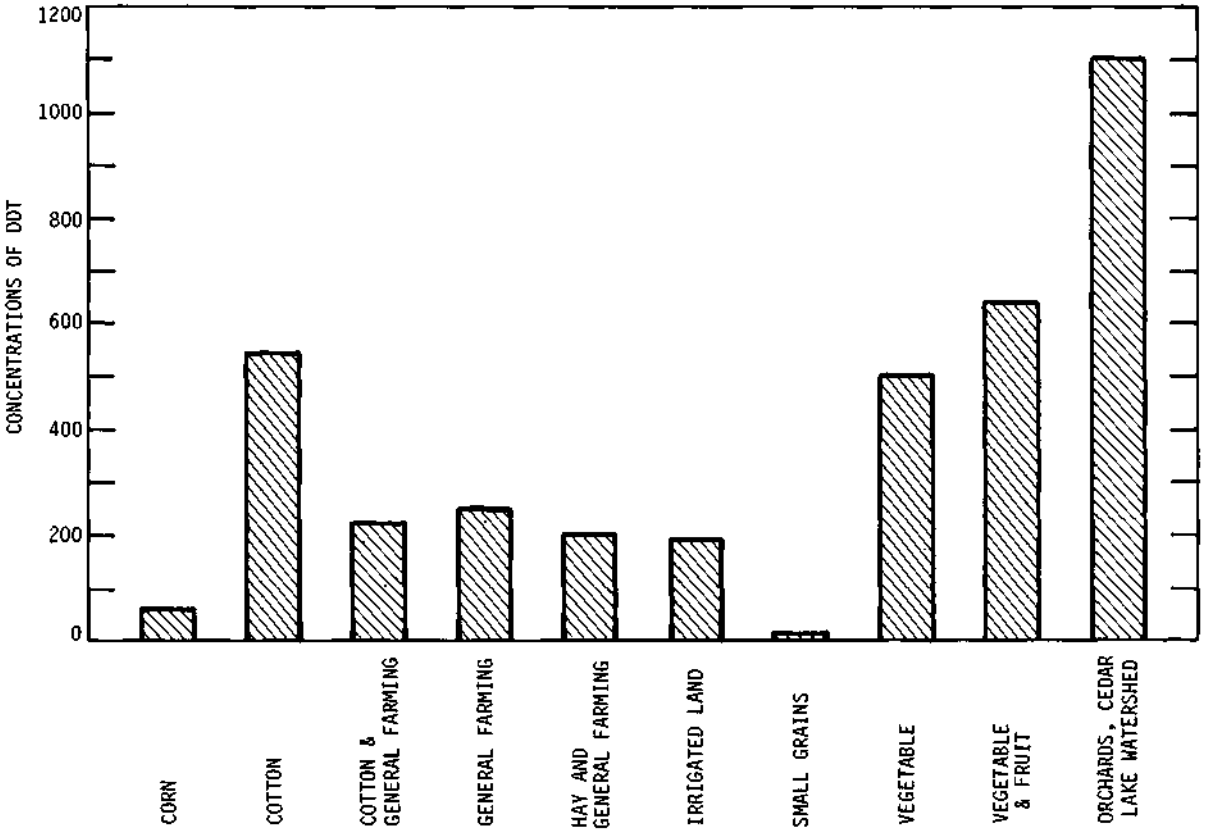

Figure 37. Comparison of DDT concentrations in soils used for different crops

Table 24. Organic Pesticides in Alluvial Sediments of Streams

(Concentrations in ppbj

				Average		
	DDT	*DDE*	*DDD*	*DDT*	*DDE*	*DDD*
Clay Lick Creek	6	30	20	9	33	20
	11	30	20			
	9	40	20			
Mill Creek	30	170	80	23	143	63
	20	110	50			
	20	150	60			
Poplar Camp Creek	6	40	30	5	40	23
	5	40	20			
	4	40	20			

Organic Pesticides in Lake Sediments

Samples were obtained from the bottom sediments and the mud-water interfaces of Cedar and Little Cedar Lakes and from the bottom sediments of the Alto Pass Reservoir. These were analyzed for organic pesticide concentrations. Organophosphates and carbamates were not found in any of the samples. In contrast to soils samples in which DDT was observed in 95 percent of the orchard soils and DDD was not detected, DDT was not found in Cedar Lake sediments but DDD existed in all lake sediments. The average concentration of those organic pesticides found in lake sediments is shown in table 25. In terms of total DDT, the metabolite DDD was the predominant form. DDT in the bottom sediments of Little Cedar Lake and Alto Pass Reservoir showed an average range of 3 to 11 ppb. These values are within background concentrations and are not unlike the average of 6.5 ppb observed in Lake Evergreen (Sievert et al., 1975), a central Illinois lake. Nevertheless, the fact that DDT exists in the bottom sediments of the 7-year old Little Cedar Lake and the 9-year old Alto Pass Reservoir suggests the concentrations are of recent origin; otherwise they would have degraded to DDD. The values, however, do not reflect a significant accumulation of DDT. The data in table 25 do show increases in the accumulation of DDT and its metabolites commensurate with the lakes' aging process. Dieldrin was not recovered in any of the lake sediments except when a florisil separation column was employed. Sievert et al. (1975) found an average concentration of 10.1 ppb of dieldrin in a central Illinois lake with a watershed consisting, predominantly, of row crop land. In 39 bottom sediment samples of a lake in California, Keith and Hunt (1966) detected a *trace* of dieldrin but the average concentration of DDT and related compounds was 4400 ppb.

DDT and its related compounds were not found in the mud-water interface of Cedar Lake. On one occasion, August 24, a concentration of 0.042 jug/1 DDE was found in Little Cedar Lake. Dieldrin was detected on three different occasions in Cedar Lake at concentrations ranging from 0.039 to 0.058 μg/l, and on one occasion in Little Cedar Lake at a concentration of 0.042 jug/1. It was anticipated that the concentrations of organic pesticides at the mud-water interface would be higher than that distributed throughout the overlying waters. Since the samples were collected at times of stratification and non-stratification, it is concluded that the waters of the lakes do not have concentrations of DDT and dieldrin exceeding public water supply standards for drinking water in Illinois, i.e., 50 μg/l and 1.0 μg/1, respectively (Illinois Pollution Control Board, 1974). The overlying waters of Lake Evergreen (Sievert et al., 1975), had a mean DDT concentration of 0.02 μg/1 with a corresponding sediment concentration of 6.5 ppb, and a mean dieldrin concentration of 0.04 μg/l with a sediment concentration of 10.1 ppb. The waters of Cedar Lake in relation to the mud-water interface and sediments should behave similarly.

Summary

Background concentrations for soils on the Cedar Lake watershed ranged between 10 and 20 ppb DDT. Orchard soil contents averaged 1098 ppb DDT and 2475 ppb total DDT (DDT and DDE). The metabolite DDD was not detected in any soils regardless of use. Dieldrin was found in three orchard soils varying in concentration from 20 to 490 ppb and in two row crop soils at concentrations of 30 and 24 ppb. Endrin was detected in one orchard at 470 ppb. Residues of organophosphates and carbamates were not recovered in any soil, sediment, or water phase of the environment.

The major pathway for organic pesticides to enter a lake system is by soil movement from the target area to the aquatic system. Because of the persistence of organochlorine pesticides in the soil, they remain a potential hazard regardless of the disuse of the pesticide or the conversion of the orchard land to other uses. It is probable that a well-sodded orchard presents less of a problem, in terms of pesticide relocation, than a former orchard being used for row crop purposes. Once in a lake system the sediment-water interactions probably govern the rate of pesticide distribution within the lake waters.

DDT, DDE, and DDD were found in the alluvial sediments of three major tributaries to Cedar Lake. The contents of those sediments deposited by Poplar Camp and Clay Lick Creeks were within the range of background levels. However, the DDT in Mill Creek sediments was significantly higher with an average total DDT of 229 ppb compared with about 64 ppb for the other streams.

DDT was not observed in any Cedar Lake sediments. DDE plus DDD ranged from 12 to 50 ppb with DDD being the predominant metabolite. Some DDT was detected in

Table 25. Average Concentrations of Organic Pesticides in Lake Sediments

(Concentrations in ppb)

		DDT	DDE	DDD
1	Cedar Lake	ND	19	31
2	Cedar Lake	ND	7	8
3	Cedar Lake	ND	18	26
4	Cedar Lake	ND	4	8
5	Little Cedar Lake	9	28	63
6	Little Cedar Lake	3	24	47
7	Alto Pass Reservoir	10	120	182
8	Alto Pass Reservoir	11	70	101

Note: ND = not detected

the bottom sediments of Little Cedar Lake and the Alto Pass Reservoir averaging 3 to 11 ppb, within background levels. As in the case of Cedar Lake, the predominant form was DDD. The average concentration of total DDT was significantly higher in the sediments of the older impoundments, averaging from 74 to 100 ppb for Little Cedar Lake and from 182 to 312 ppb for the Alto Pass Reservoir. Dieldrin was not recovered from the bottom sediments of the lakes.

DDT and its related compounds were not detected at the mud-water interface of Cedar Lake; DDE was detected on one occasion in Little Cedar Lake at a concentration of 0.042 μg/1. However, dieldrin was recovered on three different occasions in Cedar Lake, ranging from 0.039 to 0.058 μg/1. It was recovered at the mud-water interface in Little Cedar Lake on one occasion at 0.042 jug/1.

Within the detection limits of the analyses performed, it seems clear that the organic pesticide applications currently practiced on orchards in the Cedar Creek watershed are not having an adverse effect on the water quality of Cedar Lake.

If degradation rates proceed in accordance with the findings of others, those more persistent pesticides (organochlorine) used in the past will continue to diminish in importance from a water quality standpoint. This assumes that major changes in land use, leading to enhanced soil movement, do not occur.

SEDIMENTATION OF LAKES

Most midwestern streams transport a considerable quantity of soil particles, especially during storm events. These sediments in transport are fine-sized material, generally smaller than 50 μ, and consist mainly of silt and clay. When streams discharge into impoundments, fully 90 percent or more of the sediment being conveyed by stream waters is trapped in the impoundment. It has been estimated that the median rate of loss of reservoir volume due to sediment is 0.6 percent per year in Illinois (Stall, 1971). The highest and lowest sedimentation rates measured by the State Water Survey in 101 reservoirs are 5.67 and 0.15 percent per year, respectively. In Illinois, the soil loss delivered to reservoirs varies from 1.3 to 2.3 tons per acre per year.

Added to the concern for loss of capacity in reservoirs is the realization that a major pathway for introducing contaminants into a water body is the movement of soil. The quantities of inorganic and organic pesticides affixed to the soils in the watershed of Cedar Lake have been documented. The question arises: What is the annual amount of soil being conveyed from the watershed to Cedar Lake? Cedar Lake, being a newly formed impoundment, does not have a history of sedimentation sufficient to measure. In an effort to derive some meaningful data that could be applied to that lake, sedimentation studies were performed on the Alto Pass Reservoir and Little Cedar Lake. The field work for these studies was performed under the direction of James Gibb of the Water Survey's Hydrology Section.

Alto Pass Reservoir

The reservoir is owned and operated by the village of Alto Pass and serves as a source of potable water for the community. It was completed in 1967 when a dam was built across the West Branch of Cedar Creek.

The earthen dam, with its top at 648.45 feet above mean sea level, rises about 41 feet above the creek bottom. Its length at the top is 445 feet with a slide slope on the upstream side of about 3 to 1. A spillway is located on the left bank of the valley, looking downstream. The spillway crest is 7 feet below the dam.

The watershed is a hilly region. It covers about 395 acres and consists mostly of orchard and woodland. The reservoir's water surface is about 10 acres and the original storage capacity was 128.1 acre-feet (41,740,000 gallons). The topography, drainage area, and general location of the reservoir are shown in figure 38.

Methods. Four transects, shown on figure 39, were established on the reservoir. Their locations were selected principally from the features of the lake shore. Soundings for water depth and sediment thickness were made from a boat by use of a taut line across the transect sections at selected intervals. A spud bar was used to ascertain the depth of the original bottom. The data obtained for each cross section were used to prepare figure 40. The original bottom elevation is shown as a dotted line labeled 1967. The top of the sediment in the reservoir is shown by a solid line with the notation 1976. The difference in elevation between the two lines is the amount of sediment that has accumulated during the 9-year life of the reservoir.

Results, On the basis of information obtained regarding sediment thickness, estimates were made of the volume of sediment that had accumulated for each of the five water segments located between the transects, dam, and upstream end of the impoundment. It was determined that sediment occupied about 20.1 acre-feet and the current volume for water storage in the reservoir is 108 acre-feet (35,192,000 gallons). This represents a loss in water storage of about 15.7 percent at an annual rate of 1.75 percent. The annual rate is about three times the median rate observed in Illinois

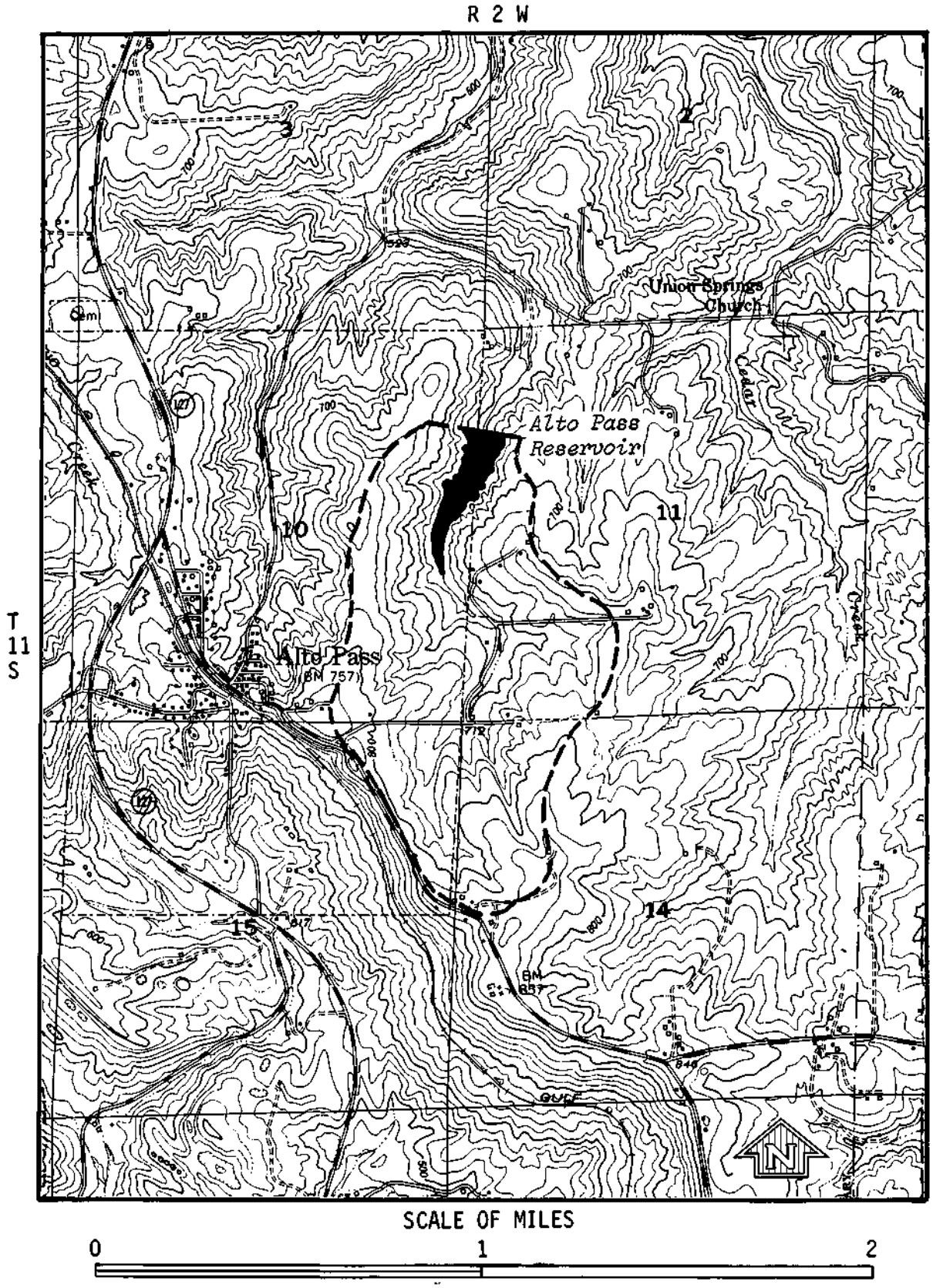

Figure 38. Location and topography of Alto Pass Reservoir watershed

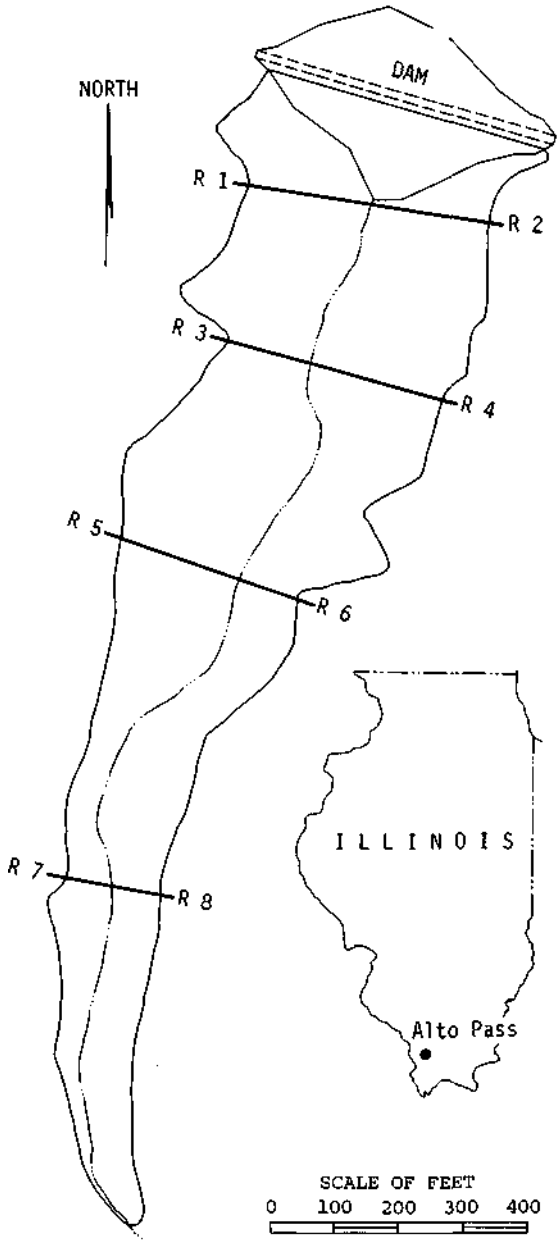

Figure 39. Sedimentation survey transects on Alto Pass Reservoir

reservoirs, even though most of the Alto Pass Reservoir's watershed is in well sodded orchards and woodlands. The upper end of the reservoir has a loss in capacity of about 42.5 percent. If we assume a weight for the sediment of 60 pounds per cubic foot, the estimated soil loss on the watershed is 7.4 tons per acre per year.

Little Cedar Lake

Little Cedar Lake, formerly known as Presley Lake, is owned and managed by the U.S. Forest Service as part of the Shawnee National Forest. It was created in 1969 to serve as a recreation base for a community development. No further development is planned for the lake and it is now maintained as a 'quiet natural area.'

The impoundment is formed by a unique arrangement in which an earthern dam has been placed across Cedar Creek and interconnected with an expanse of naturally outcropped stone which serves as a spillway. At the normal pool elevation of 437.0 feet above mean sea level, the water surface area is about 72 acres. The watershed is moderately sloping to steep and consists mainly of pastures and woodlands. The area of the watershed is about 4180 acres. The original capacity of the lake was 757.4 acre-feet (246,761,000 gallons).

The Alto Pass Reservoir is located upstream on the West Branch of Cedar Creek (see figure 3). The waters of Cedar Lake at normal elevation (431.0 msl) lap the toe of Little Cedar Lake dam. At that point portage facilities are provided to permit movement of the Cedar Lake water into Little Cedar Lake.

Methods. The methods for a sedimentation survey of Little Cedar Lake were similar to those described for the Alto Pass Reservoir. Seven transects were established on the lake. The transect locations were developed from the lake's topographical features depicted in aerial photographs and are shown in figure 41. A taut line was placed along each cross section, and soundings for water depth and sediment thickness were made from a boat at predetermined intervals. The data obtained at each transect were used to prepare figure 42. The original bottom of the lake is shown by the 1969 line, and the top of the sediment in the lake is depicted by the 1976 line. The difference in elevation between the two lines is the sediment accumulation since the dam closure.

Results. Average thickness of the sediment varied from 1 to 2 feet. From sediment measurements, estimates were made of the volume that had accumulated in each of the eight areas shown in figure 41. It was found that sediment occupied about 101.6 acre-feet and the volume of water storage now available is 655.8 acre-feet (213,660,000 gallons). This represents a depletion in water storage of about 13.4 percent at an annual rate of 1.9 percent. The depletion rate is similar to that in the Alto Pass Reservoir. Sediment distribution in the lake follows the usual pattern of deposition. In the upper end (area 6 of figure 41) the loss in water capacity is about 50 percent; in areas 1,2, and 3 the loss is about 10 percent.

The soil loss on the Little Cedar Lake watershed, assuming the Alto Pass Reservoir traps most of the sediment from its 395-acre watershed, is estimated to be 4.5 tons per acre per year. Such soil loss is about 2 to 3 times that normally expected for reservoirs in Illinois.

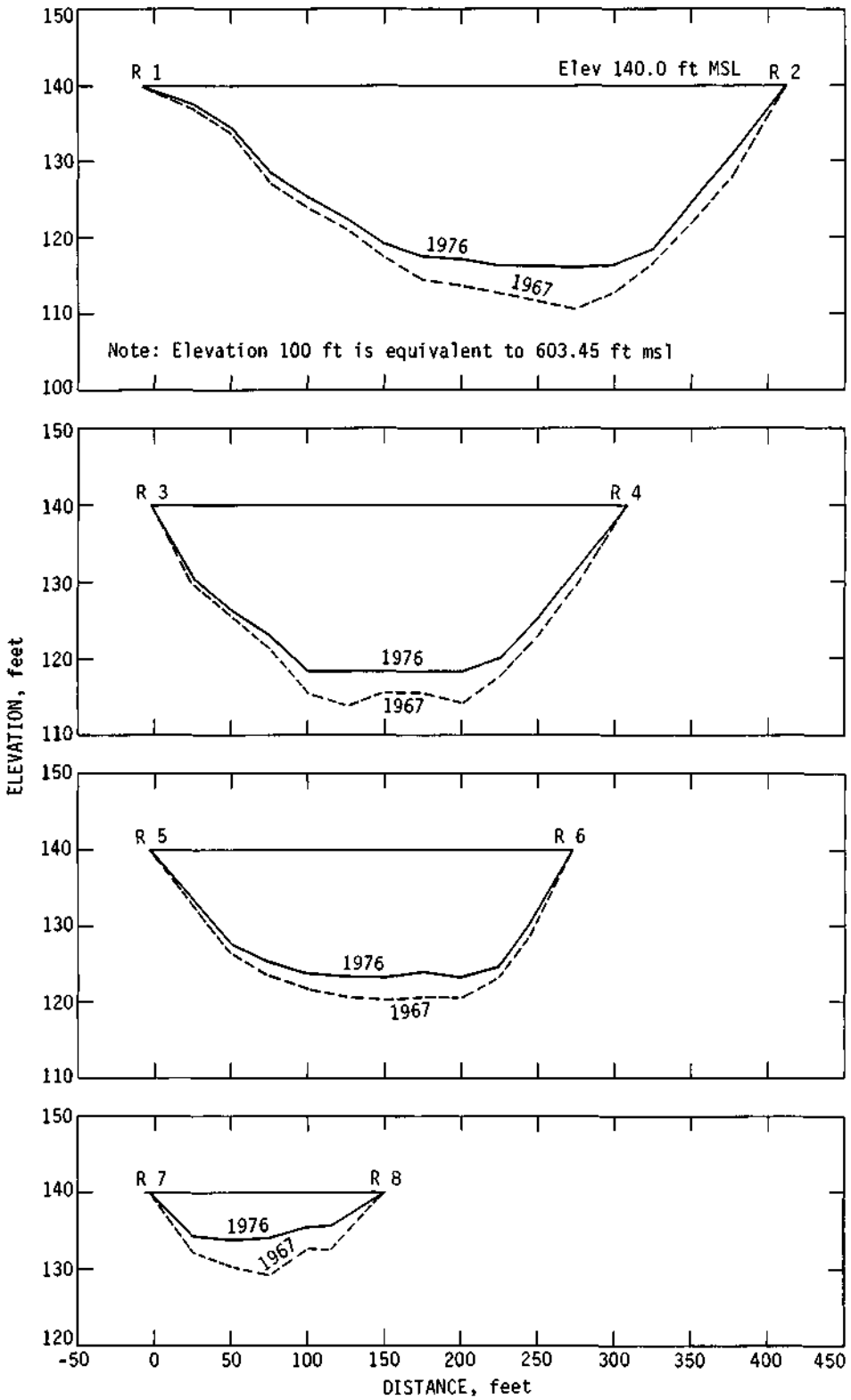

Figure 40. Depths of sediment in Alto Pass Reservoir

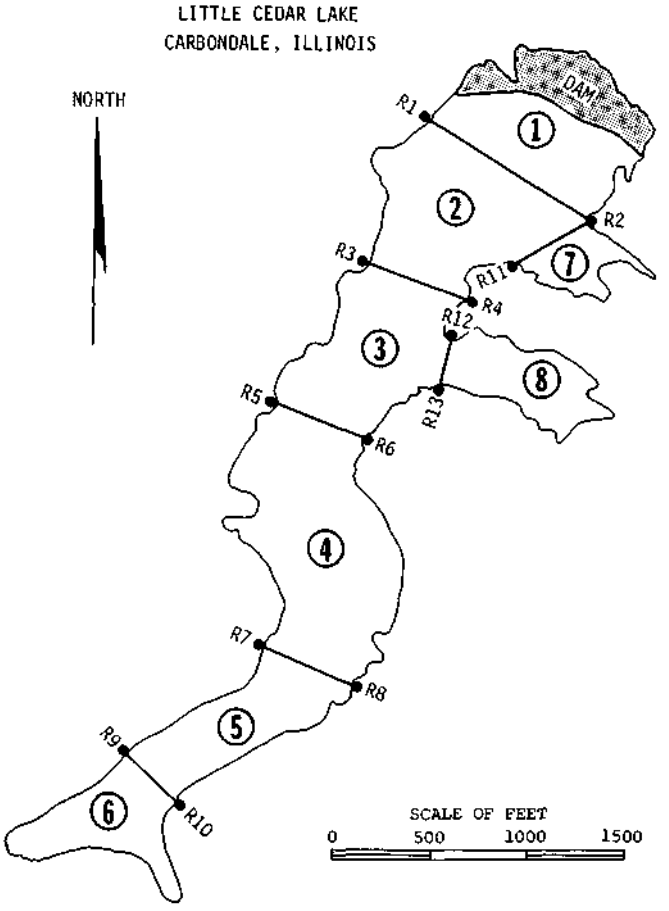

Figure 41. Sediment survey transects on Little Cedar Lake

Summary

The annual rate of water storage loss for the Alto Pass Reservoir is 1.75 percent; that for Little Cedar Lake is 1.9 percent. These losses in water storage are about three times the median rate of loss for reservoirs in Illinois. Soil loss on the watershed of the Alto Pass Reservoir is about 7.4 tons per acre per year; that for Little Cedar Lake is 4.5 tons per acre per year. Observations in other areas of the state suggest soil loss rates of 1.3 to 2.3 tons per acre per year. The exceedingly high rate of sedimentation on the watershed of the two older impoundments indicate that excessive quantities of sediment are likely to be deposited in Cedar Lake.

From the viewpoint of Cedar Lake management, it is fortunate that the Alto Pass Reservoir and Little Cedar Lake function as sedimentation traps for the Cedar Creek leg of Cedar Lake. However, the watershed area of the two smaller impoundments represents only 21.6 percent of the total watershed of Cedar Lake. If it is assumed that the remainder of the watershed will experience a rate of soil loss similar to that of the smaller impoundments, the annual sediment loading to Cedar Lake will be about 76,000 tons.

The large quantities of soil likely to be discharged into Cedar Lake, with higher than background concentrations of lead, arsenic, DDT, and dieldrin affixed to it, justifies the implementation of an effective program designed to minimize soil movement on the watershed.

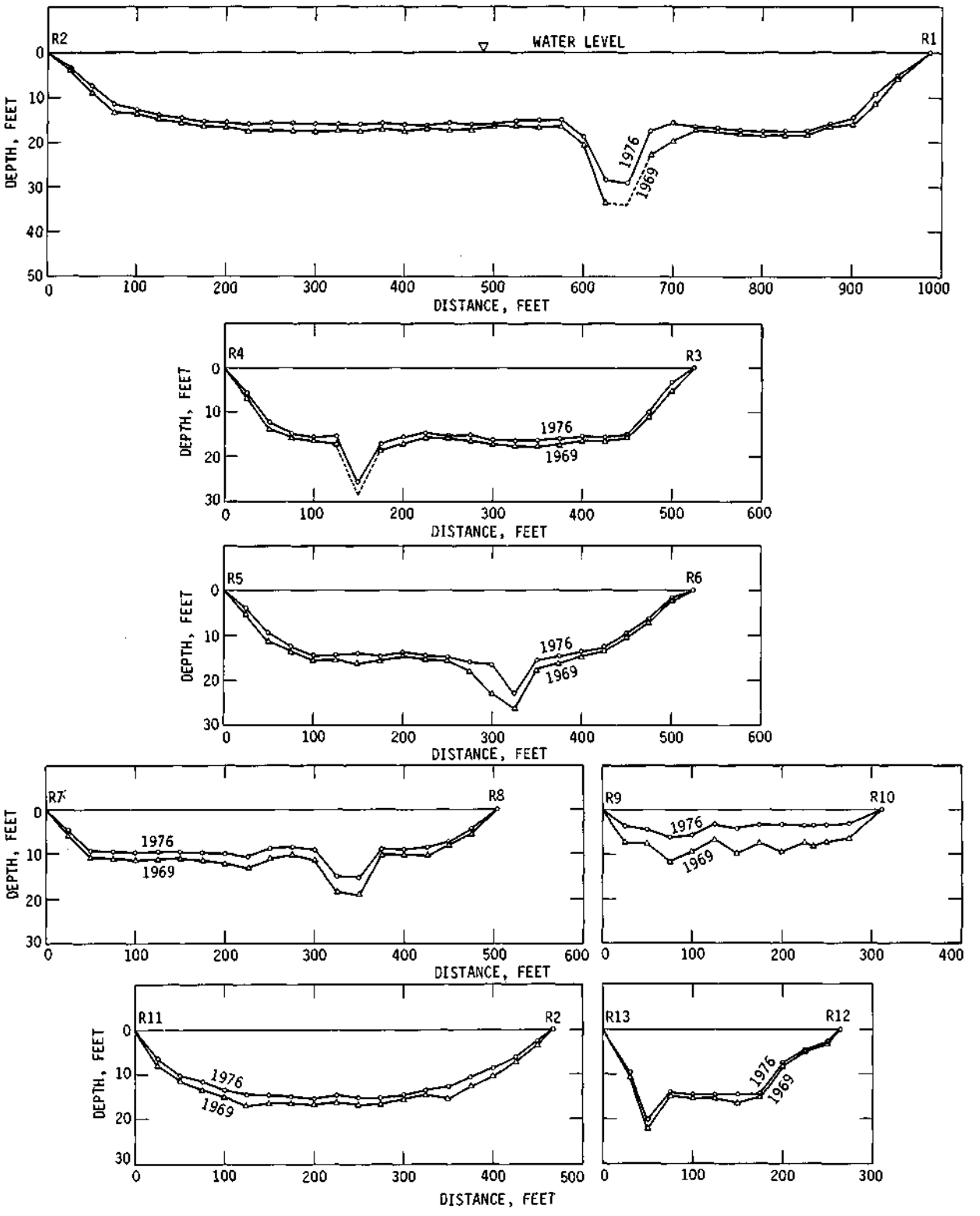

Figure 42. Depths of sediment in Little Cedar Lake

PESTICIDES IN FISH

The work plan for this study called for the collection of fish from impoundments on the Cedar Lake watershed for selective heavy metals and pesticide analyses. However, before the Water Survey field work started, the collection of fish and subsequent analyses of the fish flesh had been initiated. James Allen and Donald Garver of the Illinois Department of Conservation and Mikeal Jones of the U.S. Forest Service had obtained fish samples from Little Cedar Lake on November 18, 1975. The results of the analyses showed concentrations of mercury in the fish in excess of the tolerance limits adopted by the U.S. Food and Drug Administration (FDA). This prompted the initial concern about mercury and other toxic substances in the fishes of the Cedar Lake watershed.

Subsequent fish collections were made by personnel of the Illinois Department of Conservation (IDOC) in Little Cedar Lake and Cedar Lake. Analyses were made by the Illinois Environmental Protection Agency (IEPA) Laboratory, Springfield, and the Illinois Department of Agriculture (IDOA) Diagnostic Laboratory, Centralia. The analytical data developed from the fish collections were assembled by Robert Hite of IEPA and made available to the Survey for use in this study.

The FDA has established tolerance levels for various toxic substances in the edible portions of fish. Among them are mercury, 500 ppb; total DDT, 5000 ppb; and dieldrin, 300 ppb. Any edible portions of fish with concentrations in excess of these tolerance levels require restrictions on the quantities of fish flesh a person should consume within a specified time interval.

The pathway for the entrance of mercury, DDT and its metabolites, and dieldrin into fish flesh in the aquatic environment under Illinois conditions has not been clearly defined. Two mechanisms are often cited: adsorption directly from water and through the food chain. Both mechanisms can be generalized by the term 'biomagnification' which is defined as a process whereby the ambient concentration of a substance in the water is increased many thousands of times within the body structure of an aquatic organism. The organisms may be algae, macroinvertebrates, crustaceans, vascular plants, and fishes.

The principal purpose here is to document the concentration of mercury, total DDT, and dieldrin in some native fishes of Illinois and offer some suggestions as to why these substances are detected in fishes in excessive quantities in some water bodies and not in others.

Mercury

Mercury is not used by orchardists on the Cedar Creek watershed. Further, examinations of soils, suspended sediment, and lake bottom sediment within the watershed have not revealed concentrations of mercury significantly in excess of background levels. Yet, mercury exists in excessive quantities in several fish species in Little Cedar Lake and Cedar Lake. Similar anxiety has been voiced about certain fish species in Lake Shelbyville, Lake Carlyle, and Rend Lake. A recent study of Crab Orchard Lake, in the region of Cedar Lake, did not reveal elevated values of mercury in fishes (Hite and King, 1977).

Little Cedar Lake. The concentrations of mercury found in four species of fish collected from Little Cedar Lake on February 24, 1976, are tabulated in table 26. The species include bluegill, large mouth bass, bullhead, and carp. One bluegill and one bullhead showed concentrations of mercury in excess of the 500-ppb tolerance level. Excessive mercury was not detected in carp. However, in large mouth bass, concentrations in excess of 500 ppb occurred in 8 of the 14 samples. There is evidence to suggest that mercury concentrations in large mouth bass increase with size (Hite, 1976). For Illinois conditions it appears that elevated levels of mercury concentrations are found more frequently in predaceous fish (bass) than in bottom feeders (carp). The findings for Little Cedar Lake are consistent with that concept.

The concentrations of mercury found in fish collected from the lake on April 29, 1976, are also given in table 26. Here again excessive concentrations of mercury were detected in large mouth bass (9 of 10), but were not detected in carp. Two of the 7 bullheads collected contained concentrations above tolerance levels.

Cedar Lake. The concentrations of mercury found in four species of fish collected from Cedar Lake on May 19, 1976, are summarized in table 26. All bluegill samples (15) exhibited excessive concentrations ranging from 520 to 880 ppb. Nine of the 11 bass showed excessive mercury content ranging from 5 30 to 1800 ppb. Five of 9 bullheads showed concentrations in excess of 500 ppb. The mercury content of carp was not excessive. The pattern of accumulation of mercury in fish species is similar to that reported for Little Cedar Lake except that the bluegill population in Cedar Lake appears to be much more susceptible to mercury accumulations.

Crab Orchard Lake. Concentrations of mercury were examined in four species of fish collected from 2 locations on Crab Orchard Lake. In that 36-year old impoundment 88 fish of the four species were examined. Mercury concentrations in the fish did not exceed tolerance levels (Hite and King, 1977). The range was from 50 to 460 ppb, and means ranged from 90 ppb for bluegills to 190 ppb for large mouth bass. The authors, by extrapolation of a regression analysis for fish weight versus mercury concentration, suggest that

Table 26. Concentrations of Total Mercury in Fish

Little Cedar Lake 2/24		Little Cedar Lake 4/29		Cedar Lake 5/19	
Size (inches)	Total mercury (ppb)	Size (inches)	Total mercury (ppb)	Size (inches)	Total mercury (ppb)
Bluegill					
6.5	720	6.0	190	6.5	610
6.5	250			6.0	530
6.5	480			6.0	560
6.5	380			6.0	530
6.5	500			6.0	590
				5.5	630
				6.0	590
				6.5	520
				6.0	690
				6.5	880
				7.0	750
				6.0	590
				6.0	790
				6.5	750
				6.0	580
Large Mouth Bass					
16.5	790	17.5	1300	16.5	1800
7.5	410	14.0	520	16.5	1300
12.0	370	17.5	1450	17.0	1600
16.5	960	17.5	1650	17.0	1800
10.0	340	16.5	1590	18.0	1500
12.5	600	17.5	1890	11.5	360
14.5	560	16.0	970	10.5	500
17.0	970	13.0	430	11.0	530
8.0	400	16.0	970	11.5	540
16.5	1000	17.0	1140	12.0	555
16.5	1140			19.0	1800
17.0	1200				
10.6	270				
16.5	300				
Bullhead					
9 to 10	500	10.0	150	6.5	1200
9 to 10	340	10.5	950	7.0	430
9 to 10	700	9.5	250	7.0	530
9 to 10	400	10.5	370	6.0	230
9 to 10	200	8.0	530	6.0	360
9 to 10	320	9.5	340	6.5	520
		9.5	280	6.5	310
				13.0	1160
				12.5	1650
Carp					
3 to 5 lbs	130	18.5	290	17.0	160
3 to 5 lbs	130	20.0	60	17.0	180
3 to 5 lbs	150	19.5	140	17.0	220
3 to 5 lbs	210	18.0	180	16.0	150
3 to 5 lbs	140	19.5	80		

Note: Fishes were collected by IDOC; analyses were performed by IDOA, Diagnostic Laboratory, Centralia.

Table 27. Concentrations of Total DDT and Dieldrin in Fish, Little Cedar Lake

Size (inches)	Total DDT (ppb)	Dieldrin (ppb)
Bluegill		
6.0	360	20
Large mouth bass		
17.5	130	10
14.0	80	20
17.5	100	10
17.5	140	10
16.5	380	20
17.5	500	ND
16.0	840	20
13.0	140	20
16.0	240	20
17.0	510	10
Bullhead		
10.0	330	20
10.5	190	10
9.5	230	10
10.5	260	10
8.0	260	ND
9.5	1100	ND
9.5	420	20
Carp		
18.5	460	60
20.0	360	20
19.5	1220	220
18.0	1450	260
19.5	1530	410

Note: Fish were collected by IDOC; analyses were performed by IDOA, Diagnostic Laboratory, Centralia. ND = not detected.

Table 28. Concentrations of Total DDT and Dieldrin in Fish, Cedar Lake

Size (inches)	Total DDT (ppb)	Dieldrin (ppb)
Bluegill		
5.5 to 6.5	130	3
(Composite of 10 fish)		
4.5 to 6.5	61	5
(Composite of 10 fish)		
Large mouth bass		
11.5 to 12.5	220	28
(Composite of 2 fish)		
9.0 to 11.0	340	150
(Composite of 3 fish)		
17.5	1200	36
18.0	480	42
17.5	1820	170
17.5	380	55
17.5	350	42
Bullhead		
11.0	350	24
6.0 to 8.5	250	31
(Composite of 7 fish)		
Carp		
17.3	270	26
18.5	250	50
15.0	280	120
16.0	590	96

Note: Fish were collected by IDOC; analyses were performed by IEPA Laboratory, Springfield.

large mouth bass in excess of 6 pounds might contain mercury levels near tolerance limits.

Summary. The desirable game fish in Little Cedar Lake and Cedar Lake contain mercury concentrations in excess of FDA tolerance limits. Concentrations are not excessive in carp. On the basis of the limited collections of bluegill specimens from Little Cedar Lake compared with the bluegill population in Cedar Lake, it seems that those bluegill in Cedar Lake are more susceptible to mercury accumulations. Also, excessive mercury concentrations in bullhead seem more frequent in the newer lake system. Excessive concentrations of mercury were not detected in four comparable fish species in the 36-year-old Crab Orchard Lake.

DDT and Dieldrin

Probably the most comprehensive summary of data regarding pesticides in Illinois fish has been compiled by Barganz (1976). From his data, dieldrin is the most persistent pesticide accumulating in fish from lakes and rivers in Illinois. He also made examinations for heptachor epoxide, total DDT, and total PCB. The levels of these substances in fish were generally less than FDA tolerance levels. Pesticide concentrations appeared to be proportional to fish size. Barganz concluded that the occurrence of pesticides in fish in excess of tolerable limits is not widespread in the waters of the state.

The following discussion is confined to DDT and dieldrin levels in fish. A comparison is offered between those fish collected from the Cedar Lake watershed and fish obtained from other waters.

Little Cedar Lake. Concentrations of total DDT and dieldrin found in fish recovered from the lake on April 29, 1976, are shown in table 27. The concentrations do not exceed the tolerance limits of 5000 and 300 ppb for total DDT and dieldrin, respectively, except in one carp specimen

in which dieldrin was detected at 410 ppb. Dieldrin in other fish species (bluegill, bass, and bullhead) does not exceed 20 ppb. Though total DDT was not found in excessive quantities, its concentration in carp is more pronounced than in other fish species.

Cedar Lake. The concentrations of total DDT and dieldrin observed in four fish species collected from the lake on May 19, 1976, are shown in table 28. FDA limits were not exceeded in any fish. However, concentrations of dieldrin are generally higher for bass and bullhead in this lake than in Little Cedar Lake. The higher concentrations are not confined to carp but appear randomly distributed among all species except bluegill. Those fish from Little Cedar Lake were examined individually, while those from Cedar Lake were examined on a composite basis. The difference in procedure may account for the difference in the distribution of concentrations among the various fish species.

Crab Orchard Lake. The data developed by Hite and King (1977) were used for table 29 which includes a summary of total DDT and dieldrin concentrations observed in fish from the lake collected August 16-18, 1976. FDA tolerance levels are not exceeded in any of the four species of fish examined. The range of values for DDT and dieldrin in all fish species from Crab Orchard Lake are less than those of fish recovered in the Cedar Lake watershed.

Other Illinois Lakes. The concentrations of total DDT and dieldrin in two species of fish from other water bodies, as reported by Barganz (1976), are tabulated in table 30. The tolerance limits of the FDA are not exceeded and the range of concentrations for total DDT (10 to 140 ppb) is much more limited in value and scope than found for Little Cedar Lake and Cedar Lake. Dieldrin concentrations in carp, however, are not unlike those observed for Cedar Lake. The number of fish samples per water body are sparse for the data included in table 30, compared with the other lake systems being discussed here. The values derived, nevertheless, suggest lower concentrations of pesticides in these older water systems.

Summary. Concentrations of DDT and dieldrin in fish in excess of FDA levels are an infrequent occurrence in the waters of Illinois. Excessive dieldrin in a single carp specimen from Little Cedar Lake was the only instance reported for the Cedar Lake watershed. Of the three lakes reported on here in the southern region of the state, elevated DDT levels (greater than 1000 ppb) occur frequently in the carp population of Little Cedar Lake. The specimens collected do not approach the tolerance level of 5000 ppb in their flesh. DDT and dieldrin in fish are not a cause for concern in the Cedar Lake watershed.

Table 29. Concentrations of Total DDT and Dieldrin in Fish, Crab Orchard Lake

Size (inches)	*Total DDT (ppb)*	*Dieldrin (ppb)*
Bluegill		
6.0 to 7.5	<10-11	<10-13
Large mouth bass		
12.0 to 19.0 (Composited sample)	15-51	5-10
Bullhead		
7.5 to 14.0	<10-25	6-9
Carp		
15.5 to 19.5	74-123	19-25

Sediment Comparisons

The concentration of substances in the sediment of a lake can govern the quality of the overlying waters. This relationship of sediments has been discussed in a previous section of this report. There is a similar concept that the toxic substances found in fish are derived from the sediments. The purpose here is to compare the heavy metals and organic pesticide concentrations found in the sediments of the the lakes on the Cedar Lake watershed with that of Crab Orchard Lake. The systems are about 12 to 15 miles from each other. The data developed by Hite and King (1977) were used for Crab Orchard Lake.

Heavy Metals. Table 31 summarizes the concentrations of lead, copper, zinc, iron, and mercury detected in the sediments of the three lakes. A close examination of the data suggests there is very little, if any, difference in the concentrations. The background levels are also quite "comparable.

When Little Cedar Lake, Cedar Lake, and Crab Orchard Lake were sampled their approximate ages were 7, 2, and 36 years, respectively. It seems remarkable that their sediments are so similar. Probable exceptions to the similarity are the lead and copper concentrations in Little Cedar Lake, which appear to be higher than in the other two water bodies. The concentrations of zinc and mercury in Cedar Lake are the lowest of the three impoundments.

Organic Pesticides. There are significant differences between the sediment content of DDT and dieldrin for the lakes of the Cedar Lake watershed and Crab Orchard Lake. As shown in table 32 total DDT concentrations at background levels are significantly higher with the likely minimum concentration of 10 ppb for Cedar Lake being about three times the maximum background concentration of 3.2 ppb for Crab Orchard. Within the lake systems the concentration of DDT is significantly higher in Cedar and Little Cedar Lakes than those detected in Crab Orchard Lake.

Summary. There are only minor differences among the three lakes in terms of heavy metals in their sediments. Of basic significance, from the standpoint of mercury contamination in fish flesh, is the comparison of that element's concentration in the sediments of the lakes. Differences, if any, do not appear significant. However, the concentrations of

Table 30. Concentrations of Total DDT and Dieldrin in Fish from Selected Illinois Lakes

Location	Size (inches)	Total DDT (ppb)	Dieldrin (ppb)
Large mouth bass			
Lake Peoria	14.0	30	30
Lake Peoria	14.0	10	25
Channel and Catherine Lakes	11.5	12	<10
Fox Lake	13.0	42	10
Fox Lake	12.0	110	28
Pistakee Lake	15.5	140	<10
Pistakee Lake	16.0	80	<10
Carp			
Lake Peoria	13.0	40	32
Lake Peoria	14.0	50	80
Channel and Catherine Lakes	16.0	61	16
Channel and Catherine Lakes	18.5	63	30
Fox Lake	15.0	25	<10
Fox Lake	19.0	120	<10
Pistakee Lake	15.0	<42	<10
Pistakee Lake	18.0	79	18

Table 31. Heavy Metals in Lake Sediments

(Concentrations in ppm)

	Pb	Cu	Zn	Fe	Hg
Cedar Lake (background)	18	13	42	14,280	0.03
Crab Orchard (background)	8–15	6–12	19–35	10,000–16,000	<0.02
Cedar Lake	15–26	10–18	35–54	9,300–20,800	0.01–0.03
Little Cedar Lake	45	29–31	89–95	26,500	0.03–0.05
Crab Orchard Lake*	11–39	8–24	48–215	8,000–33,000	0.03–0.07**
	(28)	(17)	(90)	(20,600)	(0.05)

**Crab Orchard averages for 15 stations in parentheses*
***One station <0.001*

sediment DDT are much more pronounced in the watershed of the Cedar lakes than in Crab Orchard Lake.

Discussion of Causes

The concern for excessive levels of mercury in the fishes of Cedar Lake deserves a considered response. The activities required to gather the data that have been the basis of this report and their evaluation, coupled with the opportunity to review the excellent work of others, have offered some insight on the causes of mercury contamination of fish. There is convincing evidence that the source of mercury within the lake system is *not* external. There is no evidence here to substantiate the conjecture that mercury in inorganic sediment material is a major source of mercury in fish flesh. The mercury levels in Crab Orchard Lake sediments are not unlike those observed in Cedar Lake. Yet levels of mercury in the fish of Cedar Lake exceed FDA tolerance levels, and those of fish in Crab Orchard Lake do not.

The concept of biomagnification, from the viewpoint of increasing mercury levels through the food chain mechanism, does not seem logical for the waters of Cedar Lake. Benthos are scarce; algal densities are sparse. From a productivity standpoint it is an immature lake. The most likely avenue for the entrance of mercury into fish flesh in Cedar Lake is sorption.

In attempting to derive a logical explanation for high levels of mercury in the fish of Cedar Lake we can ask: What does Cedar Lake have in common with other lake systems in Illinois reflecting similar mercury contamination of their fish (i.e., Lake Shelbyville, Lake Carlyle, and Rend Lake)?

Table 32. Organic Pesticides in Lake Sediments

(Concentrations in ppb)

	Total DDT	*Dieldrin*
Cedar Lake (background)	10-20	ND
Crab Orchard (background)*	<0.4-3.2	0.40-1.5
Cedar Lake	12-50	ND
Little Cedar Lake	74-100	ND
Crab Orchard Lake	0.40-41	<0.8-34

**7 of 11 stations were less than 0.40 ppb*
ND = not detected

The most obvious thing is that they are all newly created. Further, the general practice in the creation of recent impoundments has been *not to clear* the site to be flooded, leaving vegetation inundated and trees standing to enhance aquatic habitats.

Anderson and Smith (1977) reported low levels of mercury in large mouth bass recovered from Lake Sangchris in Illinois despite the fact that the waters of the lake are vulnerable to excessive accumulations of trace elements emitted from the smokestack of a power plant complex. For cooling lake construction, it is quite likely that the basin was severely cleared of vegetation before dam closure. Potter et al. (1975) reported on a study of Lake Powell (Arizona) which also serves a power plant complex "remote from major man-caused pollution sources." The mercury content in large mouth bass was significantly higher than that reported for Lake Sangchris (314 versus 70 ppb). The mean weight of the Lake Powell fish was about 1100 grams; that for Lake Sangchris was about 490 grams. At Lake Powell, submerged decomposing snakeweed had a mean mercury content of 90 ppb compared with 32 ppb for adjacent unflooded snakeweed growing on the same substrate. Mercury levels (43 to 283 ppb) in 'lake-transported debris' were significantly higher than in terrestrial plant matter. Potter et al. (1975) suggest that mercury is concentrated in some flooded plants.

Although it was not within the scope of this study to examine all the literature regarding the content of mercury in plant material, it is generally believed that the leaves of plants contain higher levels of mercury than stems or roots.

The bottom of Cedar Lake is debris laden with decaying vegetation. Sediment samples are difficult to obtain with conventional hand dredges (Ponar; Ekman) because of debris interference. Little Cedar Lake is less laden with debris but the remnants of standing inundated trees are evidence that its basin, when flooded, was not unlike that of Cedar Lake.

We see these facts: 1) the newly created impoundments harbor mercury contaminated fish, whereas the more established lake systems (Crab Orchard Lake, Lake Decatur, Lake Springfield, Lake Bloomington, and Lake Vermilion) do not; 2) mercury content of the bottom sediments of the newer lakes do not exceed that in the bottoms of older lakes; and 3) the newly created lake basins contain vast quantities of flooded and inundated terrestial plants. From these facts, it is postulated that flooded decaying plant debris is the most influential component of the mechanism responsible for high levels of mercury in Illinois fishes. The existence of high mercury levels in the fish of the 7-year-old Little Cedar Lake suggests that the natural process for mercury reduction in lakes that have been similarly closed is a long-term one.

CONCLUSIONS

1) If only well-sodded orchard land is considered a principal source of contaminants, the past and present application of pesticides in orchards does not pose a serious threat to the water quality of Cedar Lake. But a threat does exist in those orchard lands that are, or are likely to be, converted from orchard production to other uses more susceptible to soil relocation. A program should be developed to determine the uses currently being made of former orchard land in the watershed of Cedar Lake. Records should be maintained noting the conversion of any current or future orchard lands to other uses. From a soil relocation point of view, the conversion of former orchard land to row crops, realty development, or major transportation rights-of-way are of most concern. Site specific erosion prevention procedures should be implemented in each case.

2) As shown by lake sedimentation surveys, soil loss on the Cedar Lake watershed is excessive. An effective soil management plan should be devised and implemented for the watershed, and its effectiveness should be assessed by periodic measurements of sediment transport concentrations at selected sites on Clay Lick and Mill Creeks.

3) The waters of Cedar Lake stratify resulting in depletion of dissolved oxygen in the lower 15 feet of water depth during water surface temperatures of 28 to 30 C. To minimize taste and odors and the introduction of undesirable metals in the Carbondale water supply a program should be undertaken to incorporate within the lake system devices that will minimize dissolved oxygen depletion in the Poplar Camp leg of the lake. Current technology suggests the installation of a mechanism designed to enhance complete mixing of the water body within the Poplar Camp leg.

4) A gap exists in the understanding of the distribution of arsenic and dieldrin in the aquatic environment of Cedar Lake. Arsenic, particularly in fish, should be documented.

A program should be undertaken to determine the arsenic content in the bottom muds, mud-water interface, and fishes of Little Cedar and Cedar Lakes. A more refined technique than employed in this study should be used to assess the dieldrin content in soils of the watershed.

5) A research program in cooperation with the owners of other water bodies experiencing high levels of mercury in fishes should be undertaken to determine the effects, if any, of inundated vegetation on the transfer of mercury from water to fish.

REFERENCES

American Public Health Association, American Water Works Association, and Water Pollution Control Federation. 1971. *Standard methods for the examination of water and wastewater.* American Public Health Association, Inc., 1740 Broadway, New York, N.Y. 10019, 875 p., 13th edition.

Anderson, W. L., and K. E. Smith. 1977. *Dynamics of mercury at coal-fired power plant and adjacent cooling lake.* Environmental Science and Technology v. VII:75-80.

Barganz, R. M. 1976. *Pesticide and PCB concentrations in Illinois fish.* Illinois Environmental Protection Agency, Springfield (unpublished).

Baumgartner, D. J., D. W. Schults, S. E. Ingle, and D. T. Specht. 1977. *Interchange of nutrients and metals between sediments and water during dredged material disposal in coastal waters.* Management of Bottom Sediments Containing Toxic Substances, Proceedings Second U.S. Japan Experts' Meeting, U. S. Environmental Protection Agency, Corvallis, Oregon, p. 295.

Bear, F. E. 1957. *Toxic elements in soils. In* The 1957 Yearbook of Agriculture, U. S. Department of Agriculture, Washington, D. C.

Bowen, H. J. M. 1966. *Trace elements in biochemistry.* A. P. Academic Press, Inc., London.

Butts, Thomas A., Jack W. Williams, and Ralph L. Evans. 1976. *Water quality evaluation of the Rector Creek — North Fork Saline River drainage basin.* Prepared by the Illinois State Water Survey for Soil Conservation Service, U. S. Department of Agriculture, Champaign, Illinois, 95 p.

Butts, Thomas A., Ralph L. Evans, and Shundar Lin. 1975. *Water quality features of the upper Illinois waterway.* Illinois State Water Survey Report of Investigation 79, 60p.

Cleaning our environment: The chemical basis for action. 1969. Report of Subcommittee of Environment Improvement Committee on Chemistry and Public affairs, American Chemical Society, Washington, D.C.

Colbert, Billy K., Jerry E. Scott, Jeffrey H. Johnson, and R. Charles Solomon. 1975. *Environmental inventory and assessment of navigation pools 24, 25 and 26, and upper Mississippi and lower Illinois Rivers -An aquatic analysis.* U. S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Collison, Charles, and Neil F. Shimp. 1972. *Trace elements in bottom sediments from upper Peoria Lake, middle Illinois River.* Illinois State Geological Survey Environmental Geology Note 56.

Cooperative Extension Service. 1975. *Illinois pesticide applicator study guide.* University of Illinois College of Agriculture Special Publication 39, 75 p.

D'Intri, F. M. 1971. *The environmental mercury problem.* Michigan State University Institute of Water Research Technical Report 12.

Eichelberger, James W., and James J. Lichtenberg. 1971. *Persistence of pesticides in river water.* Environmental Science and Technology v. 5:541-544.

Evans, Ralph L., and Donald H. Schnepper. 1974. *An assessment of water quality in the upper Sangamon River basin.* Prepared for the Illinois Environmental Protection Agency by the Illinois State Water Survey.

Evans, R. L., W. T. Sullivan, David Hullinger, and W. C. Wang. 1974. *The water quality of 16 water supply impoundments in Illinois.* Illinois State Water Survey unpublished data.

Garcia, W. J., C. W. Blessin, G. E. Inglett, and R.. O. Carlson. 1974. *Physical-chemical characteristics of heavy metal content of corn grown on sludge-treated stripmine soil.* Agriculture and Food Chemistry Journal v. 22(5):810.

Gross, David L. 1976. Personal communication. Illinois State Geological Survey, Urbana.

Harmeson, R. H., and T. E. Larson. 1969. *Quality of surface water in Illinois, 1956-1966.* Illinois State Water Survey Bulletin 54, 184 p.

Hatch, W. R., and W. L. Ott. 1968. *Determination of submicrogram quantities of mercury by atomic absorption spectroscopy.* Analytical Chemistry v. 40:2085-2087.

Hill, D. W., and P. L. McCarty. 1967. *Anaerobic degradation of selected chlorinated hydrocarbon pesticides.* Journal Water Pollution Control Federation v. 39(8):1259.

Hincsly, T. D: 1976. Personal communication. University of Illinois.

Hite, R. L. 1976. *Mercury contamination in Cedar Lake fishes.* Illinois Environmental Protection Agency, Marion, unpublished data.

Hite, Robert L., and Marvin King. 1977. *Biological investigation of the Crab Orchard Creek basin — summer 1975.* Illinois Environmental Protection Agency Report, Springfield, 141 p.

Hullinger, David L. 1975. *A study of heavy metals in Illinois impoundments.* Journal American Water Works Association v. 67:572-576.

Illinois Cooperative Crop Report Service. 1968. *Assessor's annual farm census 1968.* U. S. Department of Agriculture, SFC31:1-29.

Illinois Cooperative Crop Report Service. 1976. *Assessor's annual farm census 1975.* U. S. Department of Agriculture, SFC38:1-28.

Illinois Pollution Control Board. 1974. *Public water supplies.* Chapter 6, Springfield, 12 p.

Jones, J. S., and Miles B. Hatch. 1945. *Spray residues and crop assimilation of arsenic and lead.* Soil Science v. 60:277-288.

Jones, J. S., and M. B. Hatch. 1937. *The significance of inorganic spray residue accumulations in orchard soils.* Soil Science v. 44:37-63.

Keith, J. O., and E. G. Hunt. 1966. *Levels of insecticide residues in fish and wildlife in California.* Transactions of North American Wildlife and Natural Resources Conference No. 31.

King, P. H., H. H. Yeh, P. S. Warren, and C. W. Randall. 1969. *Distribution of pesticide in surface waters.* Journal of American Water Works Association v. 61:483-486.

Kothandaraman, V., Ralph L. Evans, Nani G. Bhowmik, John B. Stall, David L. Gross, Jerry A. Lineback, and Gary B. Dreher. 1977. *Fox Chain of Lakes investigation and water quality management plan.* Illinois State Water Survey and Illinois State Geological Survey Cooperative Resources Report 5, 200 p.

Kothandaraman, V., and R. L. Evans. 1971. *Analysis of variations in dissolved oxygen in an impoundment in central Illinois.* Water Resources Research v. 7:1037-1044.

Kothandaraman, V., and Ralph L. Evans. 1975. *Some water quality aspects in a nascent impoundment in central Illinois.* Transactions Illinois State Academy of Science v. 68:292-303.

Kuhr, R. J., A. C. David, and J. B. Bourke, 1974. *DDT residues in soil, water and fauna from New York apple orchards.* Pesticide Monitoring Journal v. 7:314.

Lichenstein, E. P., L. P. Depew, E. L. Eshbaugh, and J. P. Sleesman. 1960. *Persistence of DDT, aldrin and lindane in some midwestern soils.* Journal of Economic Entomology v. 53:136.

Maney, J. P., J. L. Fasching, and P. K. Hopke. 1976. *A versatile and comprehensive code for the automated reduction of gamma ray spectral data.* Unpublished data, Environmental Research Analytical Laboratory, University of Illinois.

Mathis, B. J., and T. F. Cummings. 1973. *Selected metals in sediments, water and biota in the Illinois River.* **Journal** Water Pollution Control Federation v. 45(7):1573.

Moubry, R. J., G. R. Myrdal, and H. E. Halliday. 1968. *Dieldrin residue in an orchard-dairy area of Wisconsin.* Pesticide Monitoring Journal v. 2:51.

Mowry, James. 1977. *Fruit growing influence on water quality in Illinois.* Fruit Production Subcommittee Report of 208 Agriculture Task Force Problem Assessment Report. Illinois Environmental Protection Agency, Springfield.

Murphy, R. T., J. E. Faney, and E. J. Miles. 1964. *DDT residues in southern Indiana orchards soils in 1963.* Entomology Society of America v. 19:144.

Ondov, J. M., W. H. Soller, I. Olmez, N. K. Aras, G. E. Gordon, L. A. Rancitelli, K. H. Abel, R. H. Filly, K. R. Shah, and R. C. Ragaini. 1975. *Elemental concentrations in the National Bureau of Standards' environmental coal and fly ash standard reference materials.* Analytical Chemistry v. 47:1102-1109.

Palmer, C. M. 1959. *Algae in water supplies.* U. S. Public Health Service Publication 657, 88 p.

Patrick, R., and C. W. Reimer. 1966. *The diatoms of the United States, Volume 1.* Academy of Natural Sciences, Philadelphia, 688 p.

Peden, M. 1976. Personal communication. Illinois State Water Survey, Urbana.

Pionke, H. B., and G. Chesters. 1973. *Pesticide-sediment-water interactions.* Journal of Environmental Quality v. 2:29-45.

Potter, Loren, David Kidd, and Donald Standiford. 1975. *Mercury levels in Lake Powell: Bio amplification of mercury in man-made desert reservoir.* Environmental Science and Technology v. 9:41-46.

Prescott, G. W. *1962. Algae of the western Great Lakes area.* Wm. C. Brown Co., Publishers, Dubuque, Iowa, 977 p.

Prescott, G. W. 1970. *How to know the fresh water algae.* Wm. C. Brown Co., Publishers, Dubuque, Iowa, 348 p.

Procter, P. D., T. Butz, and B. Sinha. 1975. *Heavy metal* (Cu, Pb, Zn, Cd, Ni, As, Hg) *additions to surface waters, streams sediments and selected aquatic life in Meramec Park Reservoir drainage basin, Mo.* U. S. Department of Interior, Washington, D. C, Project No. A-072-Mo.

Reuther, W. 1957. *Copper and soil fertility. In* The 1957 Yearbook of Agriculture, U. S. Department of Agriculture, Washington, D. C.

Rickert, David A., Vance C. Kennedy, Stuart W. McKenzie, and Walter G. Hines. 1977. *A synoptic survey of trace metals in bottom sediments of the Willamette River, Oregon.* U. S. Geological Survey Circular 715-F.

Roseboom, Donald P., and Dorothy L. Richey. 1977. *Acute toxicity concentrations of residual chlorine and ammonia to some native Illinois fishes.* Illinois State Water Survey Report of Investigation 85, 42 p.

Roseboom, Donald P., Ralph L. Evans, and Thomas E. Hill. 1978. *Water quality in Cedar Lake and the relevance of its watershed.* Prepared by the Illinois State Water Survey for the Greater Egypt Regional Planning and Development Commission, 93 p.

Ruch, R. R., H. J. Gluskota, and E. J. Kennedy. 1971. *Mercury content of Illinois coals.* Illinois State Geological Survey Environmental Geology Note 43.

Schacht, R. A., and W. Matsunaga. 1975. *A biological investigation of the Des Plaines River from Wisconsin state line to Joliet, Illinois.* Illinois Environmental Protection Agency, Springfield.

Sievert, A. C.,T. A. Nelson, J. M. Froramonti, D. G. Gillard, G. L. Hortin, L. S. Johnson, S. W. O'Neal, S. J. Rakow, J. M. Simkins, D. Binkenholz, and M. E. Kunz. 1975. *Effects of agricultural runoff and siltation on a new recreational lake.* Transactions of the Illinois Academy of Science v. 68:369.

Smith, G. M. 1950. *The fresh-water algae of the United States.* McGraw-Hill Book Co., Inc., New York, 2nd Edition. 719 p.

Spencer, Donald A. 1967. *Problems in monitoring DDT and its metabolites in the environment.* Pesticide Monitoring Journal v. l(2):54-57.

Stall, J. B. 1971. *Testimony on erosion and sedimentation.* Illinois Pollution Control Board, Springfield.

Stanley Consultants, Inc. 1968. *Report on Cedar Creek Dam and Reservoir.* Muscatine, Iowa, 50 p.

Sullivan, William T. 1967. *Chemical composition of the mud-water interface zone, with the description of an interface sampling device.* Proceedings 10th Conference of Great Lakes Research, pp. 390-403.

Sullivan, W. T., and D. L. Hullinger. 1969. *Phosphates in Peoria Lake, a quantitative and qualitative evaluation of a nutrient in natural waters.* Transactions of the Illinois Academy of Science v. 62:198-217.

Sustar, J. F., and T. H. Wakeman. 1977. *Dredging conditions influencing the uptake of heavy metals by organisms.* Management of Bottom Sediments Containing Toxic Substances, Proceedings Second U.S. Japan Experts' Meeting, U. S. Environmental Protection Agency, Corvallis, Oregon, p. 295.

Teledyne Brown Engineering. 1972. *Pesticide usage and its impact on the aquatic environment in the southeast.* U. S. Environmental Protection Agency, Office of Water Programs, Washington, D. C, 411 p.

Terriere, L. C, and D. W. Ingalsbe. 1953. *Translocation and residual action of soil insecticides.* Journal of Economic Entomology v. 96:751.

Terriere, L. C, Ulo Kugemagi, R. W. Zwick, and P. H. Westigord. 1966. *Persistence of pesticides in orchards and orchard soils. In* Organic Pesticides in the Environment, R. F. Gould (ed.), American Chemical Society, Washington, D. C, pp. 263-270.

Tiffany, L. H., and M. E. Britton. 1951. *The algae of Illinois.* University of Chicago Press, 407 p.

Transactions of the Illinois State Horticulture Society and the Illinois Fruit Council. 1963-1974. State of Illinois, Springfield.

U. S. Department of Health, Education, and Welfare. 1962. *Public Health Service Drinking Water Standards.* Washington, D. C, 61 p.

Vollenweider, Richard A. 1968. *Scientific fundamentals of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors of eutrophication.* Organization for Economic Cooperation and Development, Paris DAS/CST/68.27, 159 p.

Waldron, A. C. 1974. *Pesticide movement from cropland into Lake Erie.* U. S. Environmental Protection Agency, Washington, D. C. 20460, Environmental Protection Technology Series EPA-660/2-74-032.

Wang, Wun-Cheng, 1975. *Chemistry of mud-water interface in an impoundment.* Water Resources Bulletin, v. 11(4)666:175.

Wang, Wun-Cheng, and Ralph L. Evans. 1969. *Variations in silica and diatoms in a stream.* Limnology and Oceanography v. 14:941-944.

Wang, Wun-Cheng, William T. Sullivan, and Ralph L. Evans. 1973. *A technique for evaluating algal growth potential in Illinois surface waters.* Illinois State Water Survey Report of Investigation 72, 16 p.

West, P. W., and T. P. Ramachandran. 1966. *Spectrophotometrc determination of nitrate using chromotrophic acid.* Analytica Chimica Acta v. 35:317-324.

Wiersma, G. B., and H. Tai. 1974. *Mercury levels in soils of eastern United States.* Pesticide Monitoring Journal v. 7(3/4).

Wiersma, G. B., H. Tai, and P. F. Sand. 1972. *Pesticide residue levels in soils FY 1969.* National Soils Monitoring Program v. 6(3).